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Mountain farmland protection and fire-smart management jointly reduce fire hazard and enhance biodiversity and carbon sequestration

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ABSTRACT

The environmental and socio-economic impacts of wildfires are foreseen to increase across southern Europe over the next decades regardless of increasing resources allocated for fire suppression. This study aims to identify fire-smart management strategies that promote wildfire hazard reduction, climate regulation ecosystem service and biodiversity conservation. Here we simulate fire-landscape dynamics, carbon sequestration and species distribution (116 vertebrates) in the Transboundary Biosphere Reserve Gerês-Xurés (NW Iberia). We envisage 11 scenarios resulting from different management strategies following four storylines: Business-as-usual (BAU), expansion of High Nature Value farmlands (HNVf), Fire-Smart forest management, and HNVf plus Fire-Smart. Fire-landscape simulations reveal an increase of up to 25% of annual burned area. HNVf areas may counterbalance this increasing fire impact, especially when combined with fire-smart strategies (reductions of up to 50% between 2031 and 2050). The Fire-Smart and BAU scenarios attain the highest estimates for total carbon sequestered. A decrease in habitat suitability (around 18%) since 1990 is predicted for species of conservation concern under the BAU scenario, while HNVf would support the best outcomes in terms of conservation. Our study highlights the benefits of integrating fire hazard control, ecosystem service supply and biodiversity conservation to inform better decision-making in mountain landscapes of Southern Europe.

1. Introduction

Wildfires are a major component of disturbance regimes worldwide (Keeley et al., 2012). Despite the increasing amount of resources invested in fire suppression, the number of extreme fire events has largely increased over the last decades in southern Europe, overriding current fire-suppression systems (San-Miguel-Ayanz et al., 2013). In southern Europe, Spain, Greece and Portugal are the countries most affected by wildfires, both in terms of fire occurrence and total burned area (see Gonçalves and Sousa, 2017). In 2017, more than one hundred people died in Portugal due to large forest fires that overtook fire-fighting capabilities (Tedim et al., 2018). Although the total burned area in much of the Mediterranean region has decreased in recent years (Turco et al., 2016), extreme fires (i.e. large fires burning at high intensities) have become more frequent —which entails severe environmental and socio-economic impacts (Tedim et al., 2013). This increase in the number of extreme wildfires is due to increasing stand-level fuel accumulation and landscape-level fuel connectivity caused by long-standing land abandonment processes (which favor vegetation encroachment and forest densification) (Moreira et al., 2011), and more adverse climatic conditions (i.e., longer periods with hot and dry conditions, and high-speed winds) (Tedim et al., 2013; Turco et al., 2019).

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Agricultural abandonment has shaped rural mountain areas in many parts of the Mediterranean Europe since the last century, owing to diverse socio-economic and biophysical constraints such as reduced job opportunities, poor generational renewal, low accessibility and soil productivity (Cerqueira et al., 2010; MacDonald et al., 2000; Lasanta et al., 2016). The cessation of traditional livestock and agricultural practices caused by rural exodus has favored more homogeneous and flammable landscapes (Moreira et al., 2011) —with strong side-effects on fire regime, ecosystem services and biodiversity (van der Zanden et al., 2017). Rural communities traditionally used fire as a tool to manage mountain landscapes (for pastoral activities among other motivations; Chas-Amil et al., 2015; Tedim et al., 2016), being common a high frequency of low-intensity small-sized fires (Catry et al., 2009; Chas-Amil et al., 2010). The increasing amount of available fuel (under warmer and drier climatic conditions; Tedim et al., 2018) together with a high number of fire ignitions have led to altered fire regimes (Fernandes, 2013). In addition, fire-suppression policies characterized by a rigid response to fire occurrence and fire exclusion (i.e. trying to eliminate fires from the landscape) are still largely prevalent in southern Europe, indirectly contributing to foster extreme wildfires (the also-known 'Fire paradox') (Fernandes et al., 2016c). The fire-vegetation feedbacks and their complex interactions with fire-suppression policies and land-use changes make landscape dynamics difficult to predict, which challenges decision-making due to the large uncertainty of alternative management scenarios.

The progressive loss of traditional low-intensity farming systems in Southern Europe, sustaining what has been called 'High Nature Value farmlands' (HNVf, defined as socio-ecological systems underlying the maintenance of low-intensity farming systems supporting the occurrence of several species and habitats; Lomba et al., 2015), has been widely associated with population declines of many wild species (Ribeiro et al., 2014). Species associated with wet grasslands, pastures and low-intensively managed agricultural lands are the most negatively affected by land abandonment processes across Europe (e.g. grassland waterbirds and farmland bird species; Franks et al., 2018; Lehikoinen et al., 2018). On the contrary, wilderness and forest-dwelling species, including emblematic animals such as wolfs, bears or eagles, strongly benefit from land abandonment (Navarro and Pereira, 2012). This has represented a conservation opportunity in areas that became no longer viable or attractive from a socio-economic viewpoint (see 'ecological rewilding' concept, Navarro and Pereira, 2012). In relation to ecosystem services, land abandonment has boosted climate regulation (e.g. carbon storage and sequestration) (Sil et al., 2017), recreational (e.g. birdwatching or ecotourism), wood provision and water regulation services provided by forest ecosystems (Carvalho-Santos et al., 2015; Cerqueira et al., 2015; Sil et al., 2016). However, agricultural abandonment also decreases the fire regulation capacity and the fire protection ecosystem service in mountain landscapes (Sil et al., 2019b), among other ecosystem services such as food provision, pest and disease control (Renard et al., 2015).

Various fuel-treatment practices (such as prescribed burning, mechanical treatments such as forest thinning or mastication) have been proposed over the last decades to reduce fuel quantity, fuel continuity, and the associated risk of high-severity forest fires (Agee and Skinner, 2005; Omi, 2015). However, the environmental and economic sustainability of these forest practices is not always considered as they are only designed to cope with wildfires (Mclver et al., 2013). In fact, the challenge for managers and policy makers is no longer simply how to reduce wildfire impacts but how to reconcile socio-economic impacts of fires with their ecological benefits (Pausas and Keeley, 2019; Sil et al., 2019a). Fire-smart management (defined as "*as an integrated approach primarily based on fuel treatments through which the socio-economic impacts of fire are minimized while its ecological benefits are maximized*"; Hirsch et al., 2001) has emerged as an promising option to incorporate the role of fire as (socio-)ecological process into strategic planning to achieve a more sustainable coexistence with wildfires (Fernandes, 2013; Hirsch et al., 2001; Tedim et al., 2016). Fire-smart management would clearly enable a more balanced integration of positive (e.g. reducing species competition, diseases and pests or fire intensity, and increase fire protection in wildland-urban interfaces; Pausas and Keeley, 2019) and negative contributions of fire to human well-being, which would inform better decision making in fire management policy and land-use planning (Sil et al., 2019a). In practice, fire-smart landscapes can be obtained by fuel-reduction treatments and by fuel type conversion, rather than by fuel isolation (Fernandes, 2013). From this perspective, proactive management should therefore focus on reshaping vegetation (fuel) configuration to foster more fire-resistant and/or fire-resilient landscapes (Fernandes, 2013) while simultaneously ensuring the long-term supply of ecosystem services and the conservation of biodiversity (Hirsch et al., 2001; Tedim et al., 2016). To the best of our knowledge, and despite the advantages of integrated fire management for decision-making, there are no studies assessing the effects of fire-smart landscape management on ecosystem services and biodiversity conservation.

The present study aims to identify 'win-win' situations to reduce the impact of wildfires and maximize the provision of carbon storage and sequestration and biodiversity conservation in fire-prone regions affected by rural abandonment. In particular, we assessed the potential trade-offs between wildfire mitigation (measured through total burned and suppressed area), climate regulation ecosystem services (i.e., carbon storage and sequestration) and biodiversity conservation (i.e., habitat availability for 116 vertebrate species) under fire-smart management scenarios in a transboundary (Spain-Portugal) mountain region severely affected by rural abandonment and wildfires. This study illustrates the potential of wildfire-landscape dynamic model simulations to support more informed decisions for fire-suppression and landscape planning.

2. Material and methods

2.1. Study area

The study was conducted in the Transboundary Biosphere Reserve Gerês-Xurés (hereafter BR-GX) (ca. 276,000 ha, of which 71% in Portugal and the remaining 29% in Spain), a representative mountain landscape of NW Iberian Peninsula (Fig. 1). This mountainous area is covered by a complex hydrographic network running on a rugged relief, with an elevation ranging from 15 m to 1,545 m, composed of deep valleys, plains and steep slopes (Regos et al., 2015). The region is located at the transition between the Mediterranean and Eurosiberian (Temperate) biogeographic zones, close to the Atlantic coast. The study area includes the entire reserve, encompassing three EU Natura 2000 sites besides two nationally designated protected areas, the Peneda-Gerês National Park in Portugal and the Baixa Limia - Serra do Xurés Natural Park in Spain. Although our study is conducted in the entire Biosphere Reserve, we intend to discern the management impacts both within and outside protected areas (i.e. National Parks in Portugal and Spain), given the differences between both areas in terms of socio-economic values and protection measures, which would influence how the different management strategies could be implemented.

Like in other mountain areas in Mediterranean Europe, population has decreased in the BR-GX by 28% between 1990–2010, with declines of up 50% in the Spanish side (www.ine.pt and www.ine.es). This depopulation together with the ageing of the remaining farmers has been accompanied by the abandonment of traditional agricultural (with losses of approximately 10,000 hectares between 1990 and 2010) and livestock activities (including burning, grazing and extensive agriculture) in the Biosphere Reserve (Regos et al., 2015, and Appendix B) —as it has been reported for other mountain landscapes of NW Iberian Peninsula (Morán-Ordóñez et al., 2013). The landscape in the study area is dominated by shrubs (broom, gorse and heath, *c.a.* 32% of the study area) and sparsely vegetated areas (rocky areas with poor soils

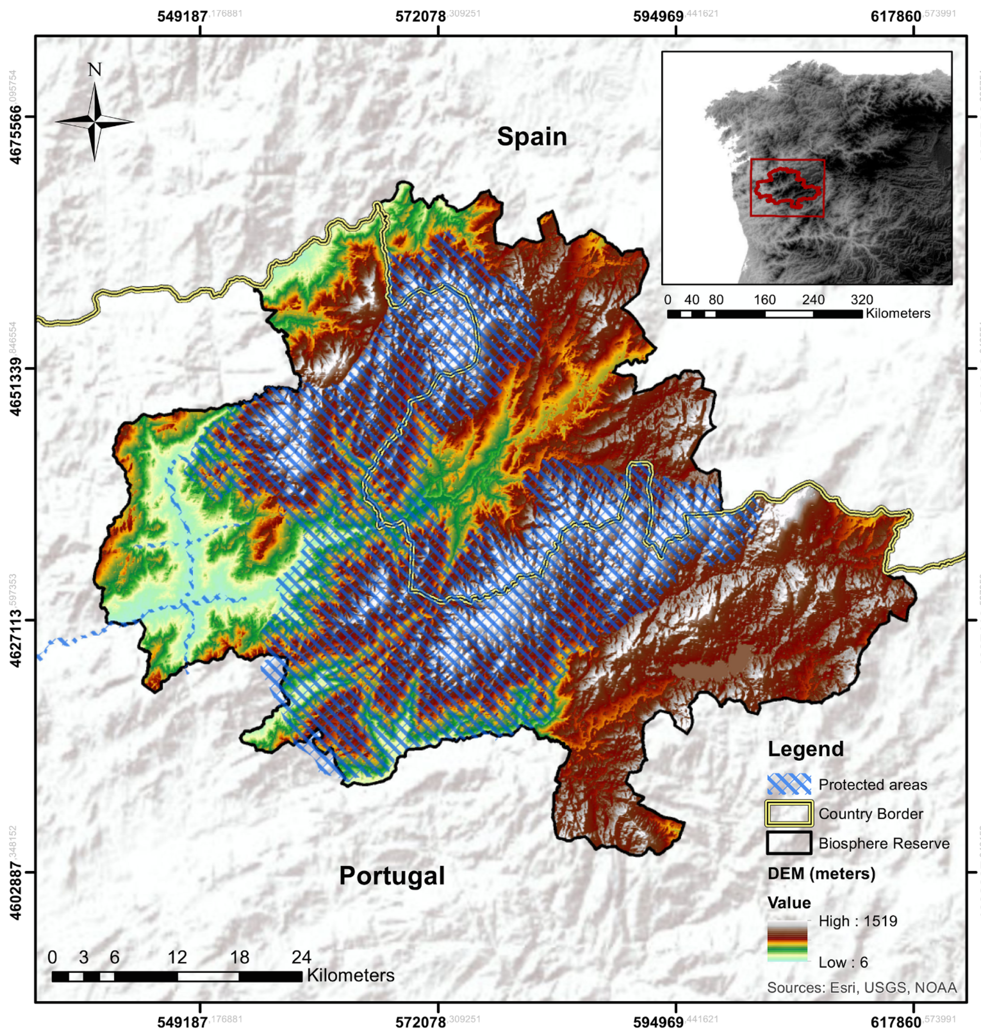

Fig. 1. Location of the Transboundary Biosphere Reserve Gerês-Xurés in the Iberian Peninsula, and the several protected areas over a digital elevation model of the area. Coordinates are UTM, Zone 29 N.

and little vegetation, 25%), maintained by fire and extensive agropastoral activities; followed by a variety of fragmented forests, such as deciduous woodlands (mostly represented by *Quercus robur* and *Q. pyrenaica*; 18%) and coniferous plantations (dominated by *Pinus sylvestris* and *P. pinaster*; 11%) (Regos et al., 2015). Deciduous oaks (the climax vegetation of the region) are less prone to fire than the more flammable pine species (see Fernandes et al., 2016a and references therein).

The study area is classified within the intermediate-cool-small pyrome (characterized by intermediate fire return but fairly small fires, see Archibald et al., 2013). Locally, the area is subjected to frequent human-induced wildfires, linked to the profound socio-economic changes suffered by these territories (e.g., vandalism, arson, revenge, land use change attempts) (Calviño-Cancela et al., 2016; Chas-Amil et al., 2015, 2010). Despite the large increase in fire-suppression resources over the last 20 years, the fire regime in the study area is characterized by large numbers of fire events and total burned area (12,755 fires between 1983 and 2010, burning a total of 195,000 ha), with areas burned up to 5–6 times (Regos et al., 2015). Between 1983 and 2010, a total of 54,041 hectares burned by around 9,512 fires in Spain. In the Portuguese side, 3,243 fires were recorded, burning a total of 141,038 hectares.

2.2. *Modelling framework and management scenarios*

2.2.1. *Workflow*

To quantify the potential impacts of alternative fire management and land-use policies on fire regime, carbon storage and sequestration and biodiversity, we coupled fire-landscape dynamic modelling with species distribution and carbon sequestration models (Fig. 2). The fire-landscape simulations allowed quantifying the impact of fire and land management on fire regime (namely, burned and suppressed area) under each management scenario, as well as the temporal dynamics of the main land cover (LC) types. Historic land-use/cover maps and the temporal projections obtained from the fire-landscape model were used as inputs in species distribution and carbon models, calibrated for past conditions (1990–2010) and projected to future landscape conditions under each management scenario (2011–2050) (Fig. 2).

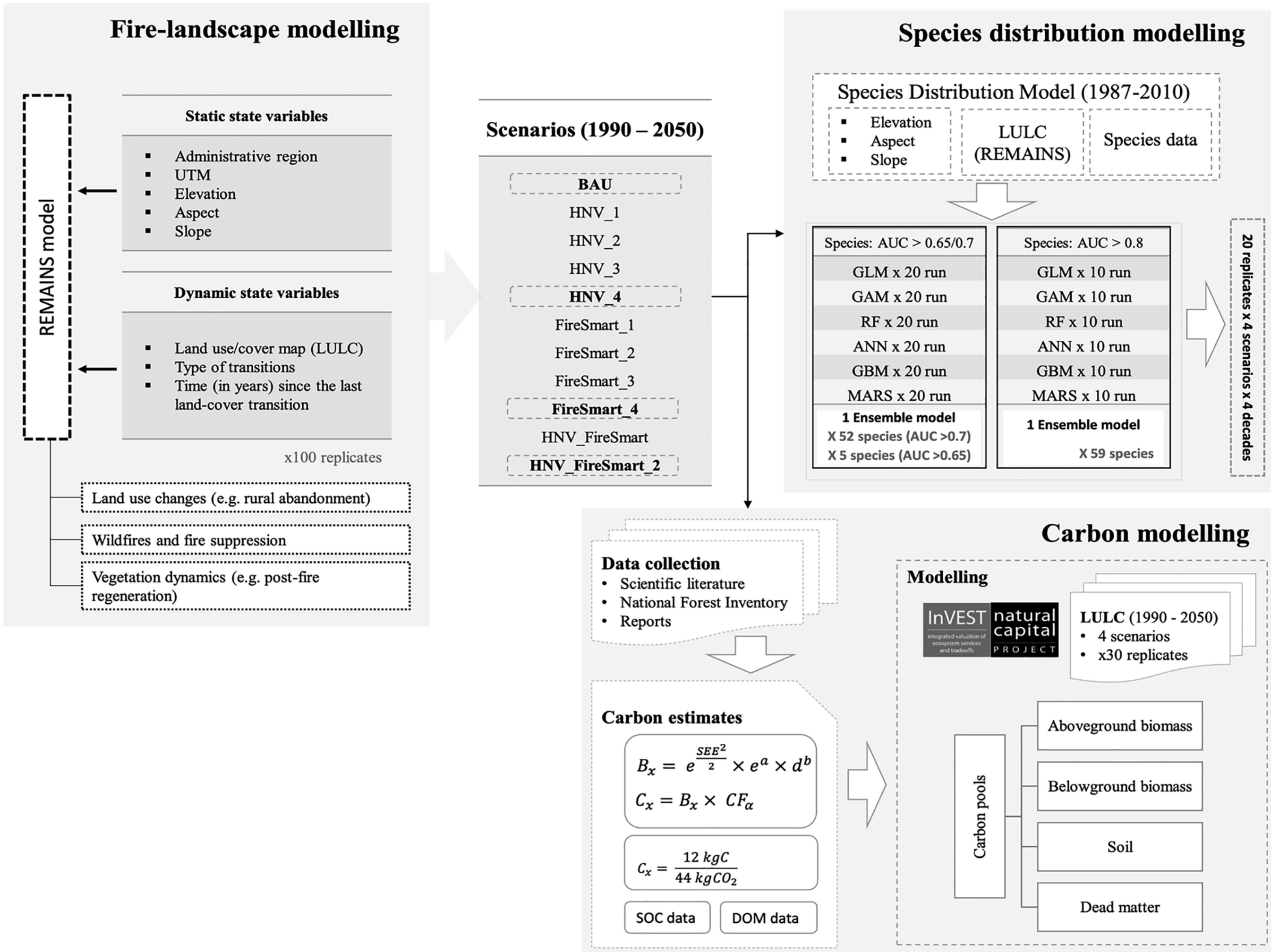

Fig. 2. Diagram of the modelling workflow including the fire-landscape, biodiversity and carbon modelling modules, the input data and the resulting outputs.

2.2.2. *The fire-landscape model*

We used a spatially explicit process-based model (REMAINS) that integrates the main factors driving fire-landscape dynamics in Southern European mountain landscapes. The model allows investigating how the spatiotemporal interactions between fire-vegetation dynamics, fire management (i.e. fire-suppression strategies) and land-use changes affect fire regime (and thereby landscape composition and dynamics) at short- and medium-timescales. It also allows quantifying the effects of fire management on total burned and suppressed area (calculated as the difference between potential area to be burnt in a year and the final burnt area). The REMAINS model reproduces fire-landscape dynamics according to pre-designed scenario storylines (Table 1). In particular, the model simulates wildfires (including fire ignition, spread, burning and extinction), vegetation dynamics (i.e., natural succession and post-fire regeneration), land-use changes (e.g. agriculture abandonment or intensification) and forest management (e.g. increase of intensive plantations for timber production) (details can be found in Appendix A).

The model was implemented using the Spatially Explicit Landscape Event Simulator (Fall and Fall, 2001), based on previous experiences using similar approaches focusing on fire-vegetation dynamics (fire-succession model MEDFIRE; Brotons et al., 2013; Duane et al., 2019) and anthropogenic land-use changes (land-use/cover change model MEDLUC (Aquilué et al., 2017). At each time step (1 year), the model

Table 1
Landscape and fire management storylines for the study area.

Name	Storyline description
Business-as-usual – BAU	It envisages a future landscape derived from the historical fire regime and land-use change trends reported between 1987 and 2010, clearly dominated by land abandonment processes (Regos et al., 2015).
High Nature Value farmlands – HNVf	Related to initiatives aimed at reverting farmland abandonment and mimicking EU environmental and rural policies on fire regime and biodiversity conservation (Lomba et al., 2015; Moreira and Pe'er, 2018) in the BR-GX as a counterpoint of the current BAU scenario.
Fire-Smart	It aims at controlling final burnt area by intervening on vegetation covers (e.g. promoting the gradual conversions of coniferous forests to native oak woodlands) to foster more fire-resistant (less flammable) and/or fire-resilient landscapes (Fernandes, 2013). Assuming the same amount of fire suppression resources applied nowadays, a more effective fire-suppression system would be expected due to lower fire spread rates found in oaks than in coniferous forests (Fernandes, 2013).
HNVf + Fire Smart	It envisages an integrated management policy that combines the promotion of more resistant and less flammable landscapes ('Fire-smart') with policies aimed at gradually increasing agricultural areas (HNVf), as an opportunity for fire suppression and farmland/grassland biodiversity conservation.

simulates fire ignition, spread and extinction until reaching a target annual burnt area defined according to statistical data for each administrative region of the study area (Portugal and Spain) between 1983 and 2010 (INCF, n.d.; MAPAMA, 2018). The target fire sizes are also a model input, but the final fire size emerges from the spatial interaction between the location of fire ignitions, landscape composition, topography, and fire suppression. The probability of fire ignition is a function of human-related and biophysical variables (see Appendix A for details). The spread rate is formulated as a polynomial expression with three factors (slope, aspect and fire-proneness of each LC type) adapted from Duane et al. (2016). Two types of fire-suppression strategies are implemented: (1) 'Active fire suppression', in which suppression of a fire front starts when the fire spread rate is below a specific threshold, mimicking the current capacity of fire brigades to extinct low-intensity fires; and (2) 'Passive fire suppression', based on opportunities derived from the presence of agricultural areas (set as 1 ha), which is assumed to be sufficient to interrupt the continuity of highly flammable vegetation, thus mimicking the advantage that fire brigades can take in heterogeneous landscape mosaics.

Land-use changes are modelled using a demand-allocation approach. In a demand-allocation framework the users set the demand (or quantity of change) and the LULC change model uses a spatial procedure to allocate the change (i.e. to select the cells to be transformed to the target land-cover type) (Aquilué et al., 2017). The demand or quantity-of-change by time step were based on a landscape change analysis performed for the 1987–2010 period (details can be found in Appendix B). Land-use/cover maps at 30-m resolution were derived from satellite images of the Landsat archive by using supervised classification methods (see details in Regos et al., 2016). In particular, four types of LC transitions are modelled per year: (1) market-oriented forest plantations, the transition of scrublands to pine plantations; (2) 'fire-smart' forest plantations, the transition of pine plantations to oak forests; (3) rural abandonment, the conversion of crops and grasslands to semi-natural vegetation areas identified as scrublands; and (4) agricultural intensification, the conversion of scrublands to cultivated land. Changes are simulated in locations with a higher likelihood to be transformed to the target land-cover type. A transition-potential variable accounts for this likelihood and is computed for each LC transition adopting the neighbor factor approach introduced by Verburg et al., (2004) (see Appendix A).

2.2.3. Fire-smart landscape management scenarios

Framed within four major storylines (Table 1), we designed 11 scenarios that combine different land management strategies in fire-prone landscapes (Table 2).

Scenario parameters regarding LC transitions were established considering historic land-use/cover changes. The annual conversion rate from cropland to scrubland (i.e. land abandonment rate) was set at 400 ha, according to the land-use change analysis between 1987 and 2010 (Table 2 and Appendix B). In 'HNVf' scenarios, the land abandonment rate was set to 0 and the annual conversion rate from scrubland to cropland was gradually increased from 400 to 1600 hectares (Table 2). The natural succession rate (value of 1.6 in Table 2) was also calculated from the land-use/cover change analysis, indicating the rate at which scrubs are converted into deciduous woodlands. This parameter was progressively increased in 'Fire-smart' scenarios to favor the conversion into oak woodlands. In 'Fire-smart' scenarios, the conversion of coniferous into deciduous woodlands were also implemented to convert half the total area or the total area of the coniferous forest area (rate of 0.5 and 1, respectively, in Table 2) into deciduous over the 40-year simulation (Table 2). One hundred replicates of each scenario were simulated for a 40-year period (2010–2050) to deal with the uncertainty associated with fire and LC transitions stochasticity.

2.2.4. Carbon sequestration modelling

We conducted a biophysical assessment of the climate regulation ecosystem service (hereafter CRES) based on the carbon sequestration ecosystem function, by applying the InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs) model to the BR-GX study area. We evaluated the impact of fire and land-use management scenarios on this ecosystem service over a period of 63 years (1987–2050). The total carbon sequestered (Tg C; i.e. the carbon sequestered by all carbon pools accumulated over time) and the total carbon sequestration rate (Tg yr^{-1}; i.e. the carbon sequestered per year by all carbon pools) were assumed as proxies of CRES.

The carbon sequestration and storage module of the InVEST model (Sharp et al., 2018) was used to perform the simulations. Uncertainty in future carbon sequestration estimates was addressed by running the InVEST carbon module for each management scenario and replicate. The carbon module links the carbon stocks in four carbon pools: Above- and belowground biomass (AGB and BGB, respectively), litter (DOM) and soil organic carbon (SOC) to each LC class type available in the study area. It returns the carbon stored in the landscape and estimates the carbon sequestered over time by comparing levels of carbon based on simulated LC spatial data. Land cover databases of the BR-GX for years 1987, 2000 and 2010 (30-m spatial resolution) and the simulated landscape scenarios (2011–2050) classified in five major LC classes (i.e., cropland, shrubland, coniferous forest, native oak woodlands, and sparsely vegetated areas) were used to feed spatial requirements of the carbon storage and sequestration module of the InVEST model. To estimate the carbon stocks in each of these pools per LC class, carbon data on AGB and BGB, DOM and SOC for each of the major LC classes was collected from: data available in published scientific literature at local or regional scale (Sil et al., 2017), and official statistics from the Portuguese and Spanish national forest inventories (Appendix C).

The carbon stocks in AGB and BGB in forest cover classes were computed based on the application of biomass allometric equations (Montero et al., 2005) to estimate the biomass available for the

Table 2
Management scenarios, their related storylines and annual LC conversion rates. Conversions from cropland to scrubland and from scrubland to oak is a natural succession process while the other two conversions are anthropogenic.

Acronym	Related storylines	Conversion from scrubland to oak (%)	Conversion from cropland to scrubland (ha)	Conversion from scrubland to cropland (ha)	Conversion of coniferous forest to deciduous woodlands (%)
BAU	BAU	1.6	400	0	0
HNV_1	HNVf	1.6	0	400	0
HNV_2	HNVf	1.6	0	800	0
HNV_3	HNVf	1.6	0	1200	0
HNV_4	HNVf	1.6	0	1600	0
FireSmart_1	Fire-Smart	1.6	400	0	0.5
FireSmart_2	Fire-Smart	1.6	400	0	1
FireSmart_3	Fire-Smart	2.0	400	0	1
FireSmart_4	Fire-Smart	2.4	400	0	1
HNV_FireSmart_1	HNVf + Fire Smart	1.6	0	800	1
HNV_FireSmart_2	HNVf + Fire Smart	2.4	0	1600	1

dominant species occurring within the area, and then converted into carbon through applying a carbon content factor (Montero et al., 2005) as shown in Appendix C. In addition, data on carbon in AGB available from the fifth Portuguese national forest inventory (http://www2.icnf.pt/portal/florestas/ifn/ifn5) was directly used after applying a conversion factor (from CO_2 equivalent to C: 12 kg C/44 kg CO_2 = 0.2727). Carbon stocks in each carbon pool for all the LC classes were maintained constant over time (assuming that carbon pools are in a steady state), which means that the carbon sequestration or emission only occurs when a pixel of a given LC type changes between dates, whereas if the LC type is kept unchanged between dates, the carbon sequestration/emission rate will be zero for that time period.

2.2.5. Species distribution modelling

We used correlative species distribution models (SDMs) to predict the impacts of alternative fire and landscape management scenarios (Table 2) on biodiversity. SDMs were calibrated using occurrence species data available from local atlas between 1990 and 2010 (Table 3). These atlases document the occurrence of breeding avifauna and herpetofauna in the Peneda Gerês National Park (Pimenta and Santarém, 1996; Soares et al., 2005) and in the Baixa Limia Xurés National Park (Domínguez et al., 2012, 2005). SDMs were performed at the spatial resolution of 1 and 2 km, depending on the atlas data used, being subsequently projected onto a 1-km grid covering the whole BR-GX for past and future environmental conditions (1990–2050) simulated under each management scenario at decanal resolution. SDMs were developed only for bird species with more than 10 presences (fewer records were available) and for herpetiles with more than 30 presences. We used a higher presence threshold for amphibians and reptiles for guaranteeing the quality of our models, as ectothermic phisiology makes modelling the distribution of these taxonomic groups potentially more challenging when using only habitat and topography as explanatory variables. Finally, we obtained suitable data for 116 vertebrate species (93, 15 and 8 species of birds, reptiles and amphibians, respectively).

The datasets from which environmental variables (topographic and land-use/cover information) were computed to calibrate the SDMs were selected according to their temporal proximity to the atlas surveys. Topographic information (altitude, slope and aspect) was obtained from the Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER) Global Digital Elevation Model (GDEM), at an initial spatial resolution of 30 m (https://asterweb.jpl.nasa.gov/gdem.asp). Land-use/cover information (percentage of coniferous forests, deciduous woodlands, agricultural areas, rocky areas with sparse vegetation, and scrubland) was obtained from (1) the abovementioned Landsat-derived maps for past conditions (1987, 2000 and 2010), and (2) fire-landscape model simulations between 2011 and 2050 under each management scenario (30-m spatial resolution). Topographic and land-use/cover information was aggregated into the spatial resolution of species data (1 km).

Habitat suitability projections were based on consensus prediction from six widely used modelling techniques available in the R package 'Biomod2' (Thuiller et al., 2009): Generalized Linear Models, Generalized Additive Models, Random Forests, Artificial Neural Networks, Generalized Models of Boosted Regression, and Multivariate Adaptive Regression Splines (Thuiller et al., 2009). The combination of different modelling algorithms in the final consensus prediction (hereafter, ensemble modelling approach) was performed using a weighted mean considering the weights proportional to the selected evaluation scores (i.e. the higher the area under the curve of the model, the greater the importance in the ensemble modelling; see Thuiller et al., 2009). This approach aims to control the uncertainty arising from individual model predictions, and to provide more informative and ecologically robust predictions (Martínez-Freiría et al., 2017, 2015; Thuiller et al., 2009). We used a repeated (at least 10 times) split-sample approach to produce predictions independent of the training data. Each model run was fitted using 80% of the data and evaluated against the remaining 20% by using the area under the curve (AUC) of a receiver operating characteristics (ROC) (Fielding and Bell, 1997). The final projection was obtained by computing a consensus of single-model projections using a weighted average approach for each vertebrate species model (Marmion et al., 2009). AUC values were considered as model weights, using at least 10 model replicates for AUC values higher than 0.65 (see Appendix D for more details). Ensemble models were converted to a binary classification of predicted presence/absence according to ROC optimized thresholds (that maximize sensitivity and specificity) from 'Biomod2' package (Thuiller et al., 2009). The total areas of habitat availability predicted by the models were summarized for groups of species defined according to different conservation/interest criteria: (1) protection under the Birds and Habitats European directives; (2) regional IUCN conservation status in Portugal and Spain; and (3) endemic species from Iberian Peninsula. For the IUCN criteria, species with status of "Least Concern" and "Near threatened" were grouped as non-threatened, while species with status of "Vulnerable", "Endangered" and "Critically Endangered" were grouped as threatened (see Appendix D for conservation status at species level). For computational efficiency, biodiversity and carbon models were only projected under scenarios with the highest contrast and most extreme fire regime and landscape trends (i.e. 'BAU', 'HNV_4', 'FireSmart_4' and 'HNV_FireSmart_2', see Table 2).

Table 3
Description of data used for calibrating the species distribution models.

Source	Sampling location	Spatial resolution	Taxonomic group	Temporal resolution	N ° Units
Atlas	PGPN	2 km	Birds	1990–1995	238
Atlas	BLXNP	1 km	Birds	1998–2000	397
Atlas	BLXNP	2 km	Birds	2010	147
Atlas	PGPN	1 km	Amphibians and reptiles	1998–2003	583
Atlas	BLXNP	1 km	Amphibians and reptiles	2010	337

3. Results

3.1. Future fire regime under management scenarios

According to our simulations, larger areas are expected to burn in the period 2031–2050 than those of historical records (1990–2010), with an annual increase of 2,000 ha, on average, under the business-as-usual scenario (Fig. 3). In contrast, alternative management policies aimed at increasing farmland areas (i.e. HNVf scenarios) would lead to a gradual reduction of the burned area in relation to the reference values (1990–2010) (Fig. 3). Indeed, the area expected to be burned by large fires (larger than 1,000 ha) between 2031 and 2050 could be potentially reduced from 20,000 ha under the BAU scenario to 10,000 ha under the HNVf scenarios (i.e. reduction of 50%; see 'HNVf_4' in Fig. 3, and Appendix E). 'Fire-smart' forest management strategies were not predicted to significantly modify fire regime in the short term (2011–2030), allowing only a significant reduction in burned area when combined with policies focused on the valorization of agricultural areas (see 'HNVf_FireSmart_2' in Fig. 3).

Regarding the effectiveness of the two fire suppression strategies, active fire suppression did not differ significantly from the BAU scenario between 2011 and 2030, even if suppressed area slightly increased from 2030 onwards (Fig. 3). In contrast, taking advantage of the opportunities created by the agricultural areas of at least 1 hectare (i.e. passive fire suppression) produced significant differences across scenarios (Fig. 3). The implementation of management policies aimed at promoting agricultural activities will likely increase future fire-suppression opportunities, leading to larger suppressed areas (from approx. 20,000 ha suppressed between 2031 and 2050 under the 'BAU' scenario up to 60,000 ha under the 'HNVf_4' scenario; Fig. 3).

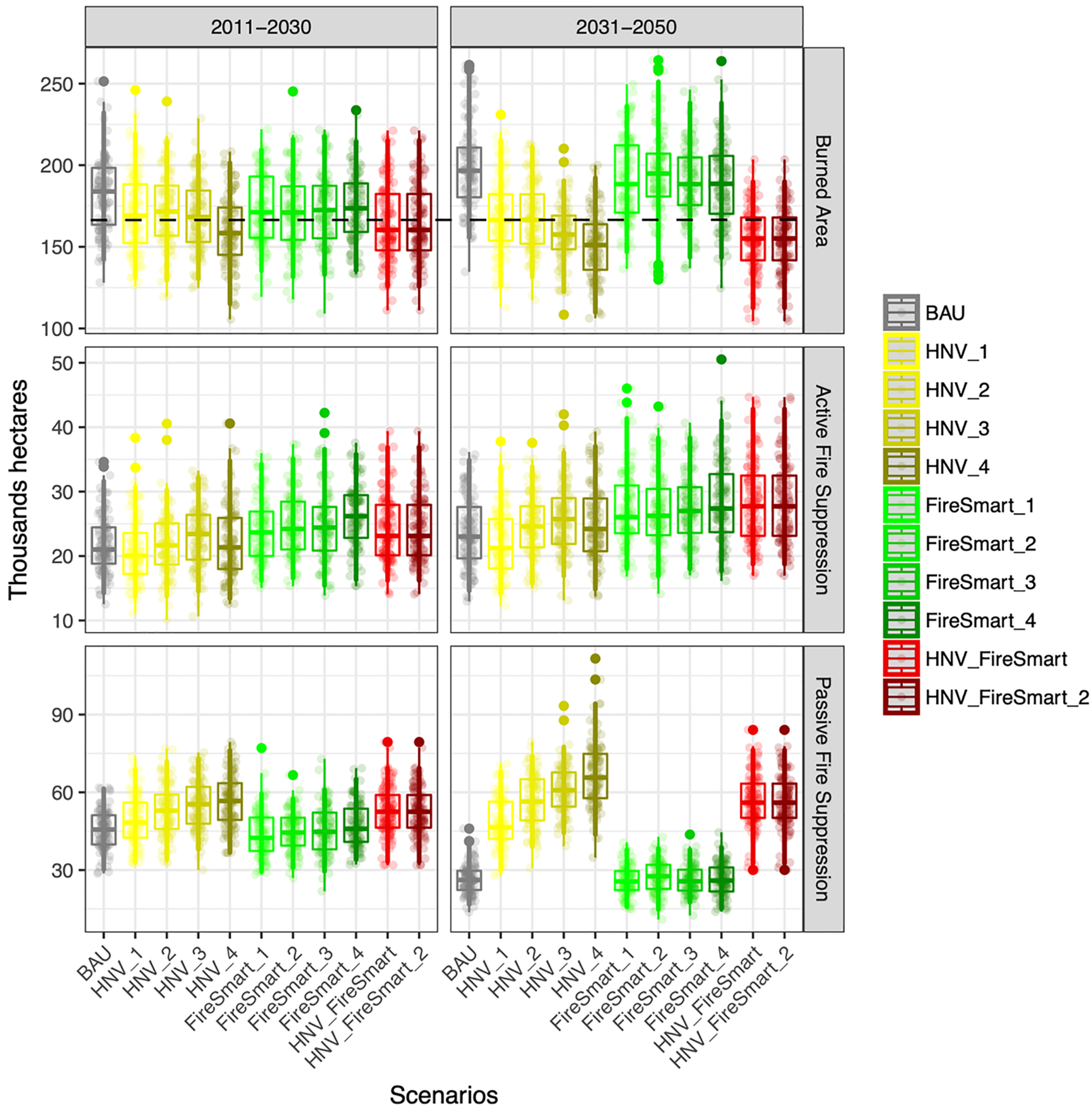

Fig. 3. Burned area and area suppressed by active and passive fire suppression strategies under each scenario in the short- (2011–2030) and medium-term (2031–2050) (see scenario acronyms in Table 2). Black line shows the area burned between 1990–2010. For all boxplots, lower and upper whiskers encompass the 95% interval, lower and upper hinges indicate the first and third quartiles, and the central line indicates the median value (solid dots are outliners). Intra-boxplot variability is computed from the different values obtained for burned and suppressed area from each model simulation, and represents the uncertainty associated with fire stochasticicy.

3.2. Carbon sequestration under management scenarios

The land-use/cover changes observed between 1987 and 2010 and simulated under each management scenario largely affected the supply of the climate regulation ecosystem service over time (measured through total carbon sequestered and carbon sequestration rate; Appendix C). Between 1987 and 2010, ecosystems in the BR-GX sequestered a total of 2.87 Tg C at an average rate of 0.12 Tg C yr^{-1}. Under the future scenarios, the results indicate that the fire-smart and business-as-usual scenarios would show the highest estimates for total sequestered carbon (4.79 ± 0.23 and 3.63 ± 0.27 Tg C, respectively) and for carbon sequestration rate (0.48 ± 0.02 and 0.36 ± 0.03 Tg C yr^{-1}, respectively), while management options aimed at promoting agricultural activities would lead to the lowest total carbon sequestered (0.27 ± 0.13 Tg C) and carbon sequestration rate (0.03 ± 0.01 Tg C yr^{-1}); still, if combined with a fire-smart strategy, higher levels of carbon sequestered would be achieved (1.23 ± 0.17 Tg C and 0.12 ± 0.02 Tg C yr^{-1}).

3.3. Biodiversity conservation under management scenarios

We obtained ensemble models with high predictive accuracy, both overall (AUC = 0.925 ± 0.07; TSS = 0.737 ± 0.167) and when

Table 4
Predictive accuracy metrics (Area under the curve – AUC; True Skill Statistics – TSS) of species distribution models according to taxonomic groups.

	AUC	TSS
Birds	0.929 ± 0.06	0.746 ± 0.168
Amphibious	0.913 ± 0.07	0.71 ± 0.156
Reptiles	0.91 ± 0.07	0.694 ± 0.169

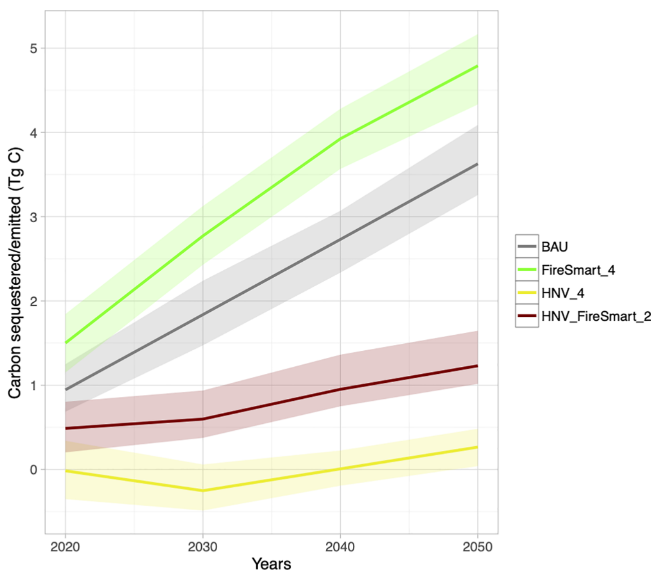

Fig. 4. Carbon sequestrated per decade between 2011 and 2050 under each management scenario (see acronyms in Table 2). For all plots, colored lines indicate mean values while the transparent colored areas indicate the error limits defined by the median range values.

considering the different taxonomic groups separately (Table 4).

Our results suggest that the business-as-usual scenario and the implementation of fire-smart strategies would lead to a steep decrease in habitat suitability (Fig. 5), independently of the protection status of the area and the species (with habitat losses varying between 10 and more than 30%, see Fig. F1 in Appendix F). Despite some variability of habitat suitability predictions, policies aimed at increasing agricultural areas (alone or in combination with fire smart strategies) are expected to largely increase, or at least avoid losses, in habitat availability for some species (Fig. 5). For species without legal protection and currently non-threatened, our results indicate an overall increase of 20–30% of suitable habitat in relation to 2010; see Fig. F.1 in Appendix F) over the next 40 years under the 'HNVf' and 'HNVf + FireSmart' scenarios (see Fig. 5). On the contrary, general decreases of habitat availability were predicted for protected (under the European directives and according to regional IUCN criteria) and vulnerable (endemic) species, both inside and outside protected areas. Nonetheless, 'HNVf' and 'HNVf + FireSmart' represent the best-case scenarios by allowing a potential habitat suitability stabilization after 2010 (see Fig. 5 and Fig. F.1 in Appendix F).

4. Discussion

This study shows the benefits of integrating proactive land-use policies and fire-smart management strategies at the regional scale (and in a transboundary context) to promote sustainable solutions to the wildfire problem in abandoned mountain landscapes. Overall, our results highlight that land-use policies aimed at promoting farmland areas would provide fire-suppression opportunities while simultaneously ensuring biodiversity conservation within (and around) protected areas. In addition, our results suggest that fire-smart management strategies based on large-scale forest conversions would foster the climate regulation ecosystem service (through carbon sequestration). This study illustrates how scenario planning supported by fire-landscape model simulations can help to better inform policy and decision making on fire and land-use management considering multiple societally relevant goals.

4.1. Sources of uncertainty and limitations

Our results confirm that the impact of wildfires in the BR-GX will be even higher in the future than nowadays under the business-as-usual scenario, due to the maintenance of rural abandonment processes (Loepfe et al., 2010) and current policies focused on fire exclusion (Moreira et al., 2019). However, these results can be relatively optimistic since climate change was not explicitly included in these scenarios. Further model developments should take into account climate-fire relationships and the complex interactions among climate, vegetation dynamics and fire management (Abatzoglou et al., 2018), given the expected increase of drought and high temperature conditions in Southern Europe (Fernandes et al., 2014; Moreira et al., 2011). It would be also worth exploring other more informative measures of wildfire impacts such as fire severity rather than total burned area, an issue especially relevant under the current context of land abandonment and climate warming.

Another potential limitation of this study is related to the assessment of a single ecosystem service —the modelling of carbon storage and sequestration. Although other ecosystem services might be indirectly evaluated, such as fire regulation capacity and fire protection services (Depietri and Orenstein, 2019; Sil et al., 2019b) or those related to biodiversity (e.g. pest control and seed dispersal; Whelan et al., 2015), we acknowledge that future studies should focus on analyzing a larger set of ecosystem services. We also recognize the potential limitation of considering broad land cover types instead of particular species or more detailed typologies. For instance, we assume that all types of croplands provide the same fire suppression opportunities, which might be an oversimplification (see e.g. HNVf typologies in Lomba et al., 2015). In addition, different species within the same shrubland and forest type could slightly differ in their post-fire response (Calvo et al., 2003, 2002), which might affect post-fire regeneration and carbon sequestration rates at local scales. Besides, while our models can estimate carbon sequestration from natural succession, post-fire recovery processes and land-use changes, direct carbon emissions from wildfires are not currently considered —an issue that should be taken into account in future model developments.

In terms of biodiversity, our species distribution models presented high predictive accuracy metrics, which allow us to interpret our results with confidence. Potential uncertainties could be associated with model downscaling (from 2 km to 1 km), but this procedure has been proven effective for capturing general environmental patterns and to predict potential distributions at finer resolutions (Araujo et al., 2005). Nonetheless, our projections to future management scenarios might still be affected by model uncertainties, particularly when considering amphibians and reptiles. Despite the application of a high threshold of presence records for these taxa to improve model quality, the ectothermic physiology of these taxa (Huey and Stevenson, 1979) coupled with the omission of climatic variables might lead to an overestimation of potential distributions. In fact, the relevance of climatic factors on shaping/limiting the distribution and dispersal of these taxonomic groups is well known, including them among the most vulnerable taxa to evaluate climate change impacts on biodiversity (Carvalho et al., 2011, 2010). As such, the predicted habitat availability for these groups (generally higher and more variable in comparison to bird species; see Fig. 5 and Fig. F.2 in Appendix F) must be interpreted with caution.

4.2. Fire regime under fire-smart management scenarios

Despite the abovementioned limitations, our simulations showed that land-use management policies aimed at promoting agricultural areas would significantly reduce the potential area burned by large fires when compared to the business-as-usual scenario (see 'HNVf' scenarios in Fig. 3, and Appendix E). These results are especially relevant given the high intensity at which large fires burn and their associated environmental and socioeconomic impacts (Fernandes et al., 2016a)

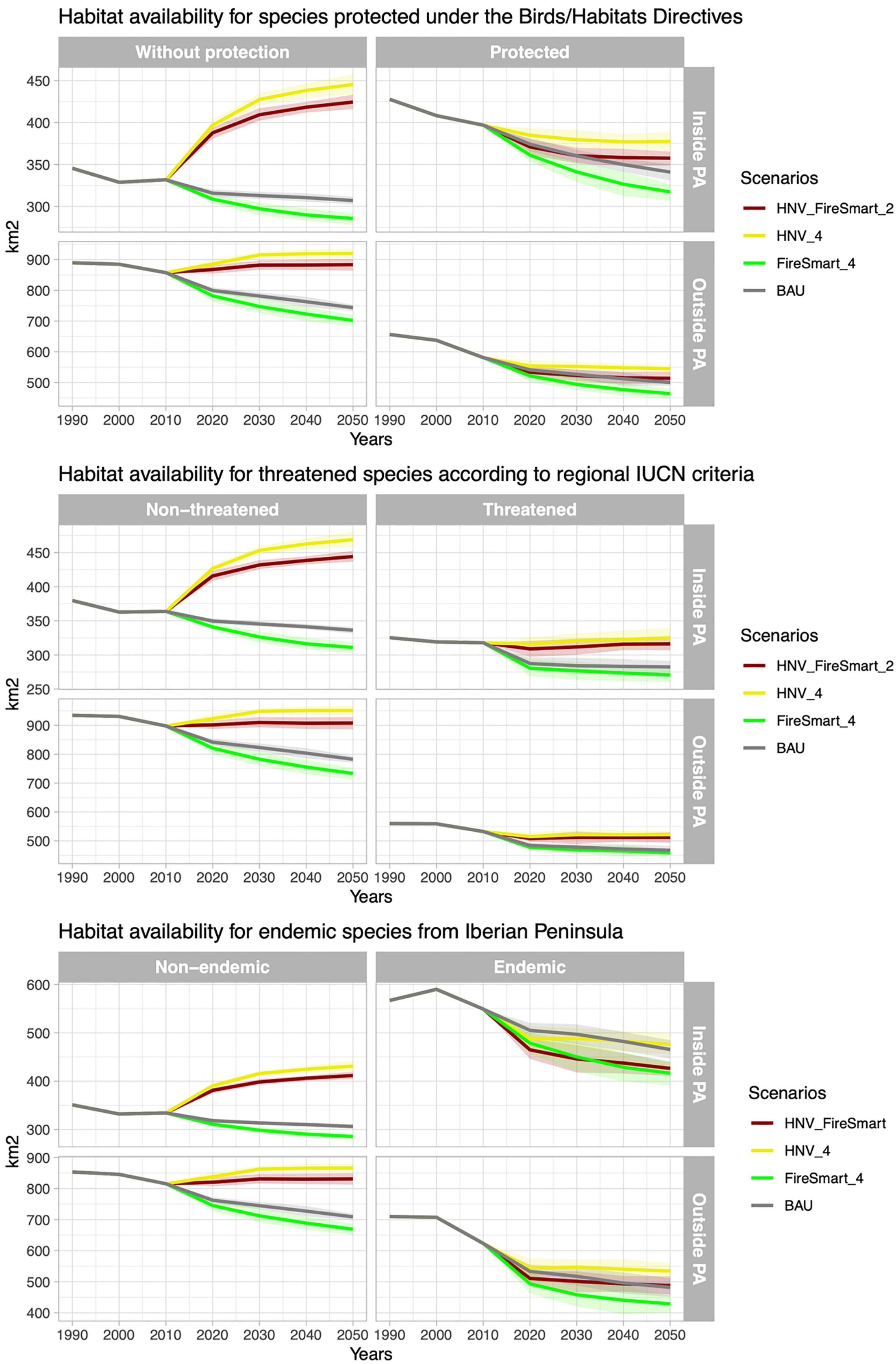

Fig. 5. Habitat availability (in km^2) for vertebrate species with and without protection status under different management scenarios within and outside protected areas (see scenario acronyms in Table 2). For all plots, colored lines indicate mean values while the transparent colored areas indicate the error limits defined by the median range values. Two protection criteria are considered: the protection under the Birds and Habitats (for amphibians and reptiles) European directives (top) and the regional IUCN conservation status in Portugal and Spain (middle). For the IUCN criteria, species with status of "Least Concern" and "Near threatened" are grouped as non-threatened, while species with status of Vulnerable, Endangered and Critically Endangered are grouped as threatened. Endemic and non-endemic vertebrates from Iberian Peninsula are also represented (bottom).

(including fire fatalities; Molina-Terrén et al., 2019). Our results are in line with recent studies suggesting that the creation of new agriculture patches (at least 100 km^2 per year in a region of 32.100 km^2) would be required to effectively reduce the total burned area in NE Spain (Aquilué et al., 2019; Moreira and Pe'er, 2018).

According to our simulations, fire-smart management strategies characterized by large-scale forest conversions to more fire-resistant forests (i.e. dominated by native oak species) would not be enough to effectively reduce potential burned area (Fig. 3). The few fire-suppression opportunities derived by the fire-smart forest conversion can be explained by the limited extent covered by forest in the Gerês-Xurés (GX) mountains (less than 15%), which undermines the capacity to affect fire regime. This might not be the case for other Mediterranean mountains dominated by large forest areas, wherein fire-smart forest conversion could be much more effective at reducing fire hazard (see review in Fernandes 2013). In our study area, the large-scale fire-smart forest conversion would be only effective in term of fire suppression if embedded in a landscape mosaic with increased agricultural areas (see 'HNVf + FireSmart' scenarios in Fig. 3). Our findings contribute to the mounting evidence that agricultural policies have a great potential to reduce future wildfire impacts (Fernandes et al., 2014; Moreira and Pe'er, 2018; Sil et al., 2019b). In light of these results, other fuel-reduction practices such as prescribed burning should be also explored in future studies given the dominance of shrublands and the traditional use of fire by rural communities to manage the landscape in northern Portugal (Fernandes et al., 2013; Úbeda et al., 2018).

4.3. Carbon sequestration under fire-smart management scenarios

Our simulations showed that land-use and fire management policies have the potential to affect the regional supply of the climate regulation ecosystem service (CRES) —measured through carbon sequestration over the next decades (Fig. 4). In particular, a higher supply of the CRES in the study area would be expected if landscape-scale forest conversion to native oak woodlands was promoted ('FireSmart' scenario in Fig. 4; see Appendix C; Roces-Díaz et al., 2017) or if the current land abandonment processes (i.e. agricultural abandonment, shrubland encroachment and native forest expansion) would continue over the coming decades (e.g. under an ecological rewilding initiative; see 'BAU' scenario in Fig. 4; Sil et al., 2017). However, an increase of fuel load and continuity favored by land abandonment processes could also lead to larger burned areas in the future (Fig. 3), with the subsequent increase in the release of CO_2 to the atmosphere (Hurteau et al., 2008; van der Werf et al., 2006).

In the absence of more efficient fire-suppression polices, our results suggest that the integration of large-scale forest conversion (from fast-growing tree plantations to native oak woodlands) within agricultural policies would promote carbon sequestration in the GX mountains with a reduced fire hazard (see 'HNVf + Fire-Smart' scenarios in Figs. 3 and 4). Nevertheless, it would be worth exploring other fuel-reduction practices (e.g. forest thinning or prescribed burning) to simultaneously reduce fire hazard and ensure the CRES under the business-as-usual scenario (Hurteau et al., 2008; Vilén and Fernandes, 2011). Management efforts either to reduce the fire hazard or promote the supply of the CRES through carbon sequestration should also consider the spatial planning of such practices by identifying priority areas where to intervene (Ascoli et al., 2018), as well as by promoting practices directed to the maintenance of an heterogeneous landscape —ensuring that future sustainability of the provision of ecosystem services and hazard reduction is achieved (Turner et al., 2013). In addition, an economic assessment of both ecosystem services (e.g. potential payments for ecosystem services) and potential economic impacts of fire on ecosystem services (either positive or negative) should be carried out to complement these results and raise awareness among landowners, managers and decision-makers (Sil et al., 2019a,b).

4.4. Biodiversity conservation under fire-smart management scenarios

Despite some differences amongst taxonomic groups, particularly for amphibians that usually select shadier vegetated habitats (e.g. forested areas) during the day due to their water/humidity dependencies (Loureiro et al., 2008), our results predicted an overall decline in species' habitat suitability over the coming decades under the business-as-usual scenario (Fig. 5), expressing rural abandonment and subsequent forest expansion (Fig. B.4). Bird species, being more habitat-specific in relation to herpetiles, respond differently depending on the scenario. According to our simulations, the bird guild most exposed to the changes of rural abandonment would be the group of species breeding in open habitats (especially for farmland and mountain species; see Fig. F.3 in Appendix F). Our predictions are in line with population declines observed for farmland and mountain bird species for Europe and North America (Lehikoinen et al., 2018; Rosenberg et al., 2019; Schipper et al., 2016). Expectedly, reptile species would benefit from policies inspired by the 'HNVf' and 'HNVf + Firesmart' scenarios, since several of the modelled species are found in – and able to adapt to – humanized habitats with extensive agricultural activities (Loureiro et al., 2008; Martinez-Freiria et al., 2019; Pleguezuelos et al., 2002). In fact, these policies might promote the heterogeneity of the landscape matrix, which might be advantageous for several reptile species by providing opportunities for thermoregulation, shelter and food availability. Our results also suggest that land-use policies promoting the expansion of farmland areas would be extremely important for the conservation of vertebrate diversity within the protected areas of the Biosphere Reserve. This management scenario is particularly advantageous for species of conservation concern (for legally protected, threatened and endemic species), since increasing farmland areas would potentially prevent a continuous and drastic loss of habitat availability in the Biosphere Reserve (Fig. 5 and Fig. F.1 in Appendix F).

4.5. Trade-offs between fire mitigation, biodiversity conservation and carbon sequestration

Our results confirm the urgent need for policies promoting farmland areas in the GX mountains, both in terms of future fire-suppression opportunities and biodiversity conservation (Figs. 3 and 5). A large amount of strategically allocated cropland areas (at least 1200 ha per year) should be gradually incorporated to the Reserve's landscape along the next decades to significantly affect fire regime in the medium term (Fig. 3). These policies would be also positive for conservation objectives since most of the species would benefit for the recovery of habitats associated with agricultural activities (Fig. 5). In terms of long-term supply of the climate regulation ecosystem service (through carbon sequestration), our models predicted the best outcomes under large-scale fire-smart forest conversion (Fig. 4). However, the integration of this fire-smart landscape conversion would be only acceptable for biodiversity conservation and fire prevention if embedded in landscape matrix characterized by increasing agricultural areas over the next decades (Figs. 3 and 5). Further studies including the socioeconomic assessment of the different scenarios and other ecosystem services such as timber production or food provision are needed to better inform decision makers on the feasibility of each management scenario.

Land abandonment has been proposed by some authors as an opportunity for biodiversity conservation in Europe (Cerqueira et al., 2015; Navarro and Pereira, 2012; Perino et al., 2019). However, when analyzed at regional and local scales, land abandonment was revealed by our results as one of the worst scenarios in terms of habitat suitability for species of conservation concern (i.e. in our case mostly open habitat species). Additionally, this process may also be the driver of more flammable and fire-prone vegetation, increasing homogeneity and landscape continuity leading to more severe fire regimes (Azevedo et al., 2011; Fernandes et al., 2016b; Moreira et al., 2011; Sil et al., 2019b). Our results are in line with other studies (see e.g. Fernandes et al., 2014) which describe similar changes in fire regime due to agricultural abandonment (Azevedo et al., 2011). This general pattern highlights the relationship between agricultural abandonment and fire hazard in the Mediterranean region (Moreira et al., 2011). Therefore, farming activities should continue to play a key role in supporting the sustainable development of rural territories as well as fire hazard management (Moreira et al., 2011; Sil et al., 2019b). In the worst-case scenario, where agricultural policies cannot be implemented in practice (i.e. ongoing land abandonment), other fuel-reduction strategies (such as prescribed burning or forest thinning operations) should be tested to explore how rewilding initiatives could be partially navigated to enhance their positive effects on carbon storage/sequestration and reduce their negative impacts on biodiversity and wildfire hazard.

5. Conclusions

This study highlights the urgency for transnational policy coordination promoting farmland areas in transboundary mountain regions of Southern Europe. Our results show how an effective implementation of agricultural policies would reduce fire hazard while simultaneously ensuring biodiversity conservation. In addition, our results suggest that, although fire-smart forest conversion strategies would be beneficial for a long-term supply of carbon sequestration, their implementation should be integrated within agricultural policies to jointly reduce fire hazard and conserve local biodiversity adapted to these semi-natural systems. Our study evidences the benefits of integrating fire hazard control, ecosystem service supply and biodiversity conservation to better inform decision-making in mountain landscapes of Southern Europe.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. . Fire-vegetation model simulations: processes, state variables, and initialization

Model purpose

The REMAINS model integrates the main landscape-level processes driving fire-vegetation dynamics in mountain landscapes of northern Iberian Peninsula. It includes the main anthropogenic and natural (abiotic and biotic) drivers of landscape change to study their spatiotemporal interactions and feedbacks effects. The model allows investigating the interlinked effects of wildfires (including fire suppression), land-use change and management strategies on fire regime and overall landscape composition. Ultimately, the model aims to generate spatially explicit scenarios of landscape dynamics according to pre-designed scenario storylines. Future landscape configurations will be used as input for subsequent biodiversity and ecosystem services model simulations.

State variables

REMAINS is supported by three state variables (updated in each model time step) and seven auxiliary variables.
State variables:

1. *LCM* (the land-use/cover map) describes the main land use/cover types of the study area: agricultural land, scrublands, rocky areas with sparse vegetation, coniferous forest, oak woodland, water and urban. All of them are considered dynamics, except water and urban that are static (Fig. A.1).
2. *TSChg* indicates the time (in years) since the last land-cover transition.
3. *TransType* records which type of transitions has more recently taken place in each grid cell: a stand-replacing fire, an anthropogenic driven transition (i.e., rural abandonment – from cropland to scrubland –, agriculture intensification – from scrubland to cropland –, forest plantation – from scrubland to coniferous forest–, and fire-smart plantation – from coniferous forest to oak woodland –), or a natural successional transition (i.e. afforestation – from scrubland to oak woodland – and vegetation encroachment – from rocky areas with sparse vegetation to scrubland –).

Auxiliary variables:

4. *AdminRegion*: the study area encompasses 2 countries, Portugal and Spain.
5. *RoadDens*: road density (used in the logistic model to predict fire ignition probability; Fig. A1).
6. *UTM*: 1 × 1 km UTM grid (used to dynamically compute ignition probability).
7. *Elevation*: digital elevation model (DEM; in m).
8. *Aspect*: aspect derived from the DEM (1 - north, 2 - east, 3 - south, and 4 - west).
9. *SlopeDegree*: slope derived from the DEM (in degrees).
10. *ProbIgni*: ignition probability (Fig. A.1).

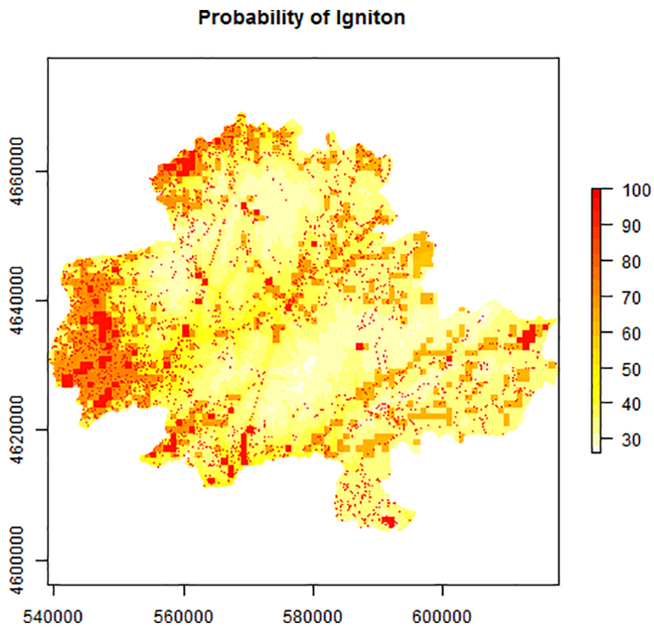

Fig. A1. Fire ignition probability layer derived from a logistic regression that includes topography (altitude and slope; source: DEM), accessibility (road density; source: OpenStreetMap) and wildland-urban interfaces (source: Landsat-derived maps) as main factors.

The spatial resolution of all spatial variable is 30 m^2 and the temporal resolution of the model is set at 1 year.

Process overview

The processes that REMAINS simulates can be grouped in three major drivers of change: (1) land-use changes (e.g. rural abandonment, agriculture conversion and coniferous plantations), (2) wildfires and fire suppression, and (3) vegetation dynamics such as post-fire regeneration, vegetation encroachment and woody colonization of scrublands (i.e. afforestation) (Fig. A2).

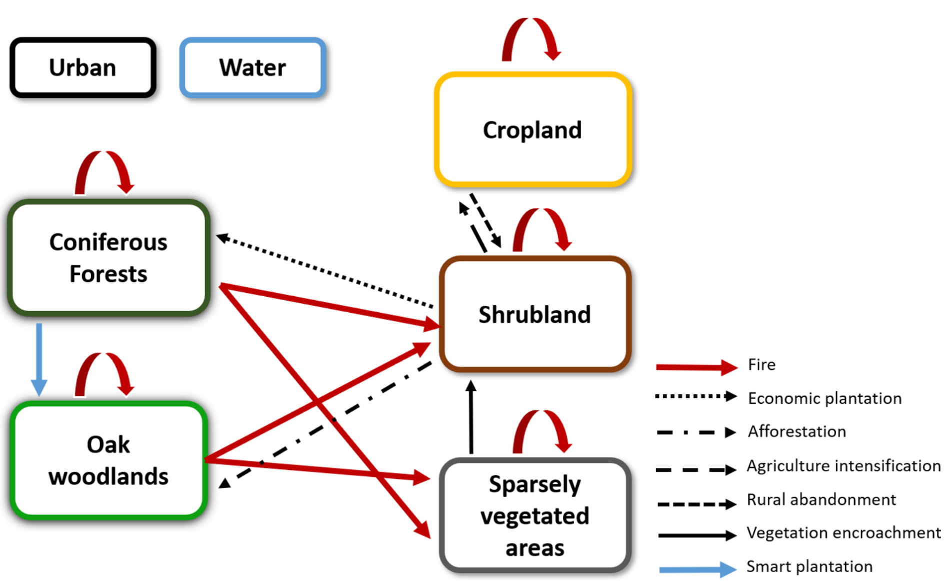

Fig. A2. Land-cover types for the Gerês-Xurés Transboundary Natural Park and both anthropogenic and natural driven land-cover transformations. Potential post-fire state is indicated in red for both the dynamic and the static land-cover types.

I. Land use changes

In the current version of the model, and to answer the proposed research questions, four anthropogenic-driven land cover transitions are modelled: (1) market-oriented forest plantations, the transition of scrublands to coniferous plantations, (2) 'fire-smart' plantations, the transition of coniferous plantations to oak woodlands, (3) rural abandonment, the conversion of crops and grasslands to semi-natural vegetation areas identified as scrublands, and (4) agriculture intensification, the conversion of scrublands to cultivated land (Fig. 1). In our modelling framework a land-cover transition (e.g. rural abandonment) is defined by a unique target land-cover (e.g. scrublands) and all the land-covers that may undergo change (e.g. croplands and grasslands). Land-cover transitions are modelled using a demand-allocation approach. The demand or quantity-of-change by time step has to be provided by the user (based on e.g. historical trends or to emulate landscape scale management policies). Changes will occur in locations with a higher likelihood to be transformed to the target land-cover. We initialized the likelihood for each land-cover transition as the neighbourhood factor of the target land-cover introduced by (Verburg et al., 2004). The neighbourhood factor of a land-cover accounts for the relative abundance of such land-cover within the neighbourhood and the overall landscape. By initializing the likelihood of conversion by the neighbourhood factor we assume that land-cover transitions tend to homogenize the landscape, by happening where the target land-cover is already more abundant. The allocation of the changes follows an algorithm that recognizes the emergence and contagion character of such processes (Aquilué et al., 2017).

II. Wildfires and fire suppression

Fires are the main stand-replacing natural disturbance currently included in REMAINS. A given landscape implicitly has a fire regime associated, even though it may be altered or modified by extreme climatic conditions (Duane et al., 2019). We adopted a top-down approach to model the fire regime at the landscape level (that currently does not depend on climate). At each time step, each fire event ignites, spreads, and stops to reach a predefined target annual area that is specific for each administrative region of the study area. We used fire statistical data (obtained from the Spanish Ministry of Agriculture and Fisheries, Food and Environment) between 1983 and 2010 at the municipal level, and data in shapefile format for Portugal (obtained from the Institute of Nature Conservation and Forests - ICNF) for the period 1990–2010. From these data, annual burnt areas were quantified to create a burnt area distribution and thus incorporate historical fire regime into the landscape-fire model. The target fire sizes are a model input too, but the final fire size emerges from the spatial interaction between the location of the fire ignition, the landscape composition, the topography and the fire suppression when applied. The probability of fire ignition is a function of human-related and biophysical variables. In the current version, fire ignition is modelled as a multivariate logistic regression model where explanatory variables are elevation, density of roads, and neighbourhood configuration that is described at 1 × 1 km cells (matching the UTM grid).

The logistic regression model:

$$logit\,(P_{ignition|non-ignition}) = \textit{-0.01613} + \textit{0.002047}\cdot\text{RoadDens} - \textit{0.001321}\cdot\text{Elevation} + \textit{1.987}\cdot\text{UrbNat} + \textit{1.568}\cdot\text{CrpNat}$$

being ‘UrbNat’, the interface between urban and natural areas; and ‘CrpNat’, the interface between cropland and natural areas, that are dynamically updated in each time step.

Thus, the spatial distribution of fire ignitions depends on landscape configuration, elevation and accessibility, while fire spread depends on slope, aspect, and land-cover flammability. The spread rate is formulated as a polynomial expression with three factors (slope, aspect and fire-proneness of each LCT) adapted from Duane et al. (2016).

Final fire sizes can be explicitly reduced if a fire suppression strategy (or a combination of strategies) is activated. Currently two firefighting actions are designed: (1) a fuel-based strategy that take advantage of fire spreading situations bellow a specific fire spread threshold to suppress the fire and (2) a landscape-based strategy that uses open mosaics to stop the advancing fronts. Fuel-based threshold is the maximum spread rate at which fuel-based suppression can be activated and the landscape-based is the number of contiguous agricultural lands (1 hectare) that have to burn before the landscape-based suppression strategy can be activated. These are scenario parameters and when are set at 0 suppression is not activated (Table A1).

Fire strategy defined at the administrative level (counties) according to the calibration procedure:

III. Vegetation dynamics

For simplification, we assume that natural vegetation in mountain landscapes always follows the same successional pathway, from rocky areas to closed scrublands, and then to oak woodlands. The transformation from one state to the following only occurs after a fixed period of time since the last transition and depends on the presence of potential colonizers in a circular neighbourhood around the target location.

Fire spreads and burns all land covers except urban areas and inland water, these are oak woodland, coniferous plantations, scrublands, rocky or open scrublands, grasslands and agricultural lands. After fire, agricultural lands, scrublands and open scrublands persist (i.e. there is no land-cover change because of fire) while forest stands may partially change state to scrublands or to rocky scrublands (Fig. 1). The variable ‘time since last change’ (proxy of the biomass) always turns to 0. A percentage of forest areas (coniferous and oaks) return to the pre-fire state after some years after fire while some forest areas remain scrublands or rocky vegetation for a while (Table A2).

1. Afforestation is the colonization of scrublands by oak species, occurs at a certain annual rate (2.6%), only can take place after a time period since the last land-cover transformation (9 years) and it depends on the percentage of oak woodland found in a circular neighbourhood of radius (13 cells, approx. 400 m) around the target location.
2. Vegetation encroachment is the colonization of open vegetation areas by scrublands, occurs at a certain annual rate (2.2%), only can take place after a time period since the last land-cover transformation (4 years) and it depends on the percentage of scrublands in a circular neighbourhood of radius (4 cells, approx. 120 m) around the target location.
3. Post-fire recovery: non-forest areas (agricultural land, scrublands and open vegetation) persist after fire, that is there is no post-fire transition for these land-covers. For coniferous forests and oak woodlands, the post-fire regeneration is stochastic, even if a percentage of the regeneration occurs by contagion, that is mimicking the post-fire pathway undertake by a close burnt neighbour. However, when any of these forest species are set to persist (i.e. remain coniferous or oaks, respectively) these take a certain time (i.e., number of years) to effectively recover the forest canopy (Table A.3). In order to estimate the average post-fire recovery profiles for each land cover class, the normalized difference vegetation index

Table A1
Firefighting effectiveness levels for each administrative region and fire suppression strategy.

Administrative Region	Fuel-based Th (%)	Landscape-based Th (ha)
Spain	50	1
Portugal	30	1

Table A2
The post-fire probability for each LC class.

pre\post	crop	shrub	coniferous	oak	rckyveg
crop	100	0	0	0	0
shrub	0	100	0	0	0
coniferous	0	19	42	0	39
oak	0	13	0	54	33
rckyveg	0	0	0	0	100

Table A3
Post-fire recovery rates: % pixels and time to recover to the pre-fire LC class (expressed in years).

Initial state	Final state	Transition	Rate
Coniferous	Coniferous	42%	8 years
	Rocky	39%	
	Shrub	19%	
Deciduous	Deciduous	53%	9 years
	Rocky	33%	
	Shrub	13%	
Shrub	Shrub	77%	4 years
	Deciduous	11%	9 years
Rocky	Rocky	78%	4 years
	Shrub	13%	4 years
Cropland	Cropland	100%	1 year

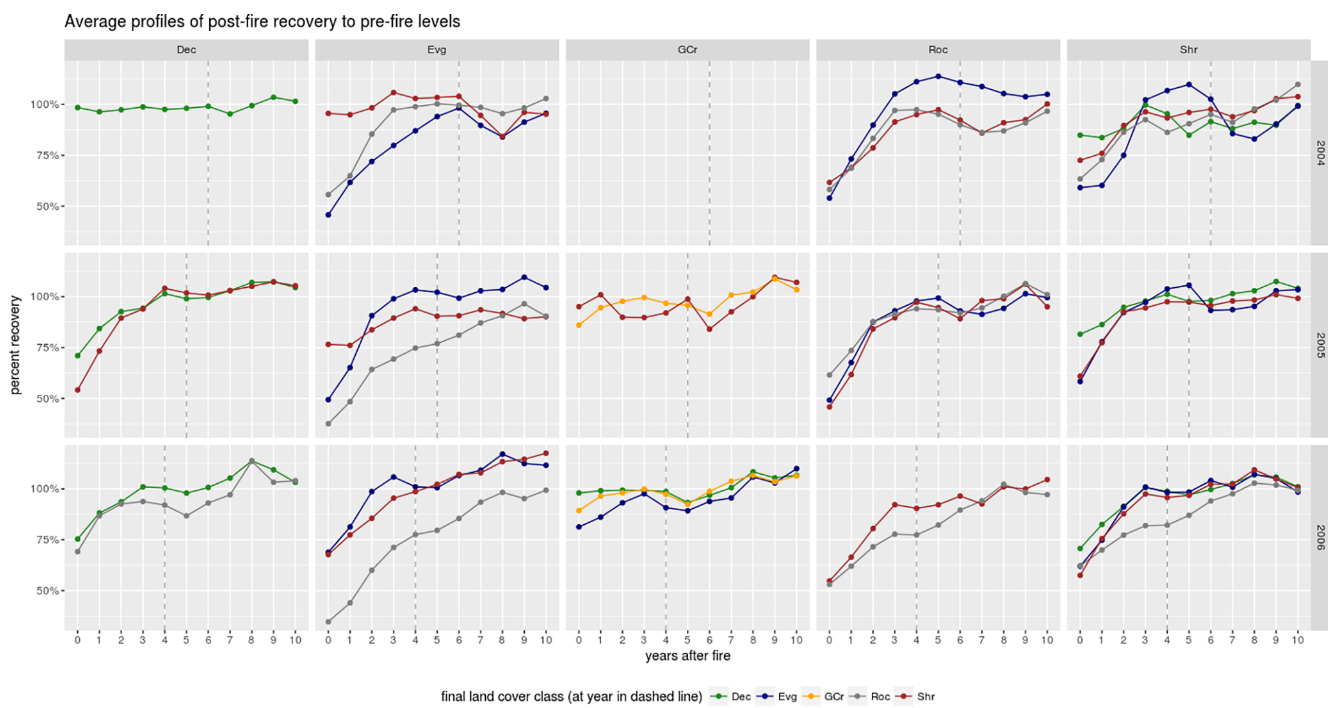

Fig. A3. Average profiles of post-fire recovery to the pre-fire levels for each land cover class computed from fraction of vegetation cover (FVC).

(NDVI) was extracted from the MODIS Vegetation Indices product (MOD13Q1, Collection 6, 250 m, 16-day). Burned area masks were extracted from the MODIS Burned Area Product (MCD64A1, Collection 6, 500 m, monthly) for the years 2001–2016. All MODIS image time-series were re-projected to WGS84/UTM29N reference system, and resampled to 250 m, using nearest neighbour method. Then, the NDVI time-series was used to calculate an approximation of the fraction of vegetation cover (FVC), by applying the formula in Gutman & Ignatov, 1998:

$$FVC = ((NDVI - -NDVI_{min})/(NDVI_{max} - -NDVI_{min}))^2$$

where $NDVI_{min}$ and $NDVI_{max}$ correspond to values for bare soil (FVC = 0) and dense vegetation (FVC = 1), respectively. The values of this NDVI-based FVC were then used to extract post-fire regeneration curves for each pixel in the study area identified as burned in each of the years 2004, 2005, or 2006, but not in any of the preceding years (from 2001). Median post-fire regeneration profiles, with values for the 10 years following the fire year, were then calculated from the aggregation of pixels with a percentage area of each land cover class in 2000 equal or greater than 60%, as well as for each relevant land cover class transitions between 2000 and 2010 (Fig. A.3).

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Appendix B. . Land-use/cover analysis and model simulations.

To analyze land-use/cover changes at landscape level, we cross-tabulated the remote sensing data-derived maps in order to obtain a transition matrix for quantifying the spatial extent that had been lost or gained by a given land cover type over the entire study period (from 1990 to 2010) (Fig. B.1).

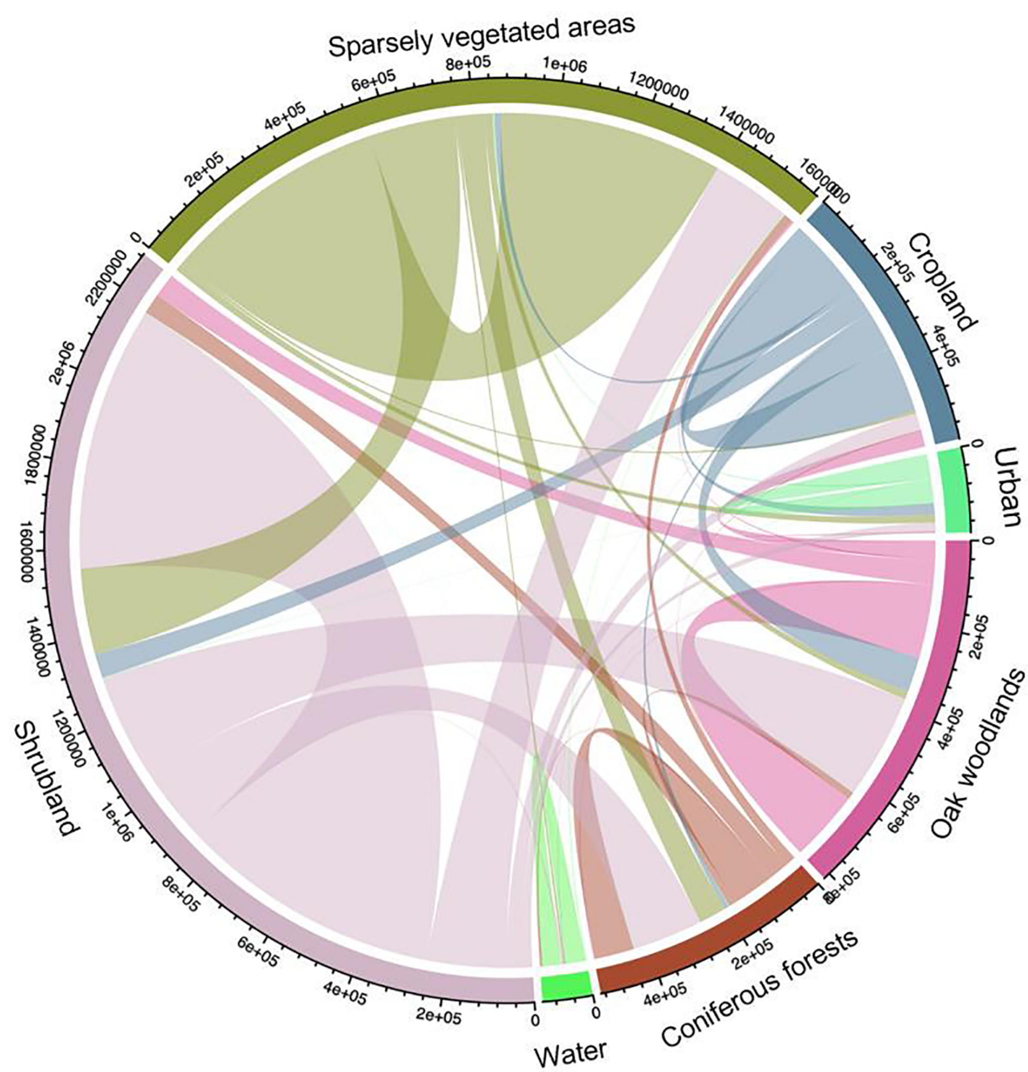

Fig. B1. Circular plot illustrating the land-cover type transitions between 1990 and 2010, in hectares (ha). The size of the lines is proportional in width to the contribution of each land-cover type to the change. The colors refer to the land-cover types.

During model calibration exercises, different values of the different model parameters were tested until the reference values were achieved. For instance, different afforestation and natural succession rate values (from the initial values obtained from the remote sensing analyses) were tested by running model simulations of the landscape from 1987 to 2010 under the business-as-usual scenario until the total amount of each LC type were achieved (Fig. B.2).

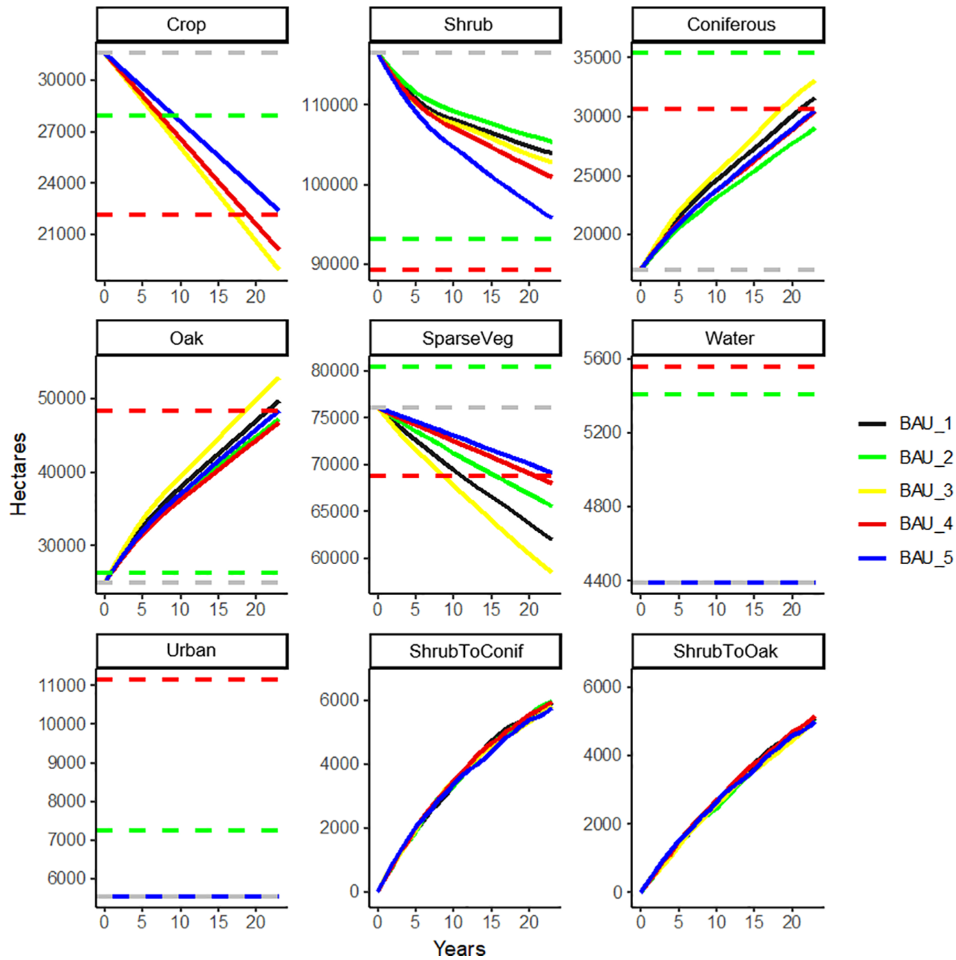

Fig. B2. Example of a model calibration exercise. Model simulation from 1987 to 2010 under different natural succession and afforestation rates estimated from historic LC trends. Red color line represents the target values to be achieved, represented by hectares for each LC type in year 2010.

Model simulations were carried out under each management scenario to predict the amount of each LC class since 2010 to 2050, at annual timescale. Model outputs are available in raster spatial (Fig. B.3) and table format (Fig. B.4).

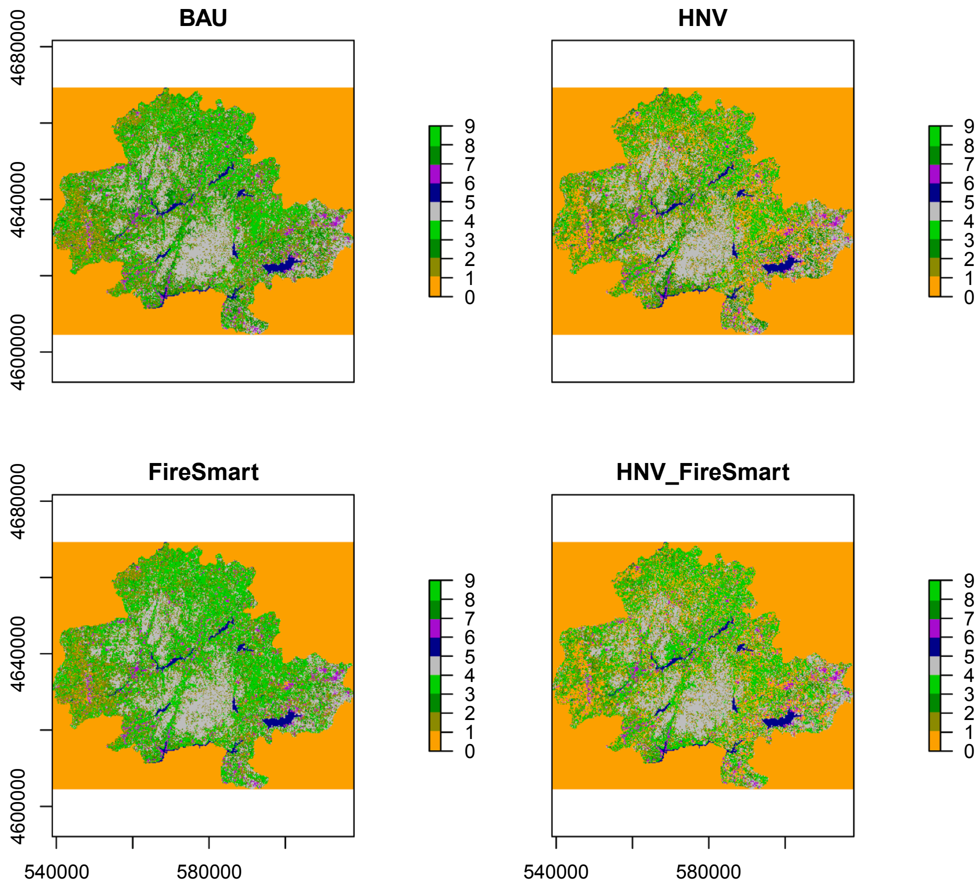

Fig. B3. Spatial representation of one of the 100 model simulations under different future management scenarios.

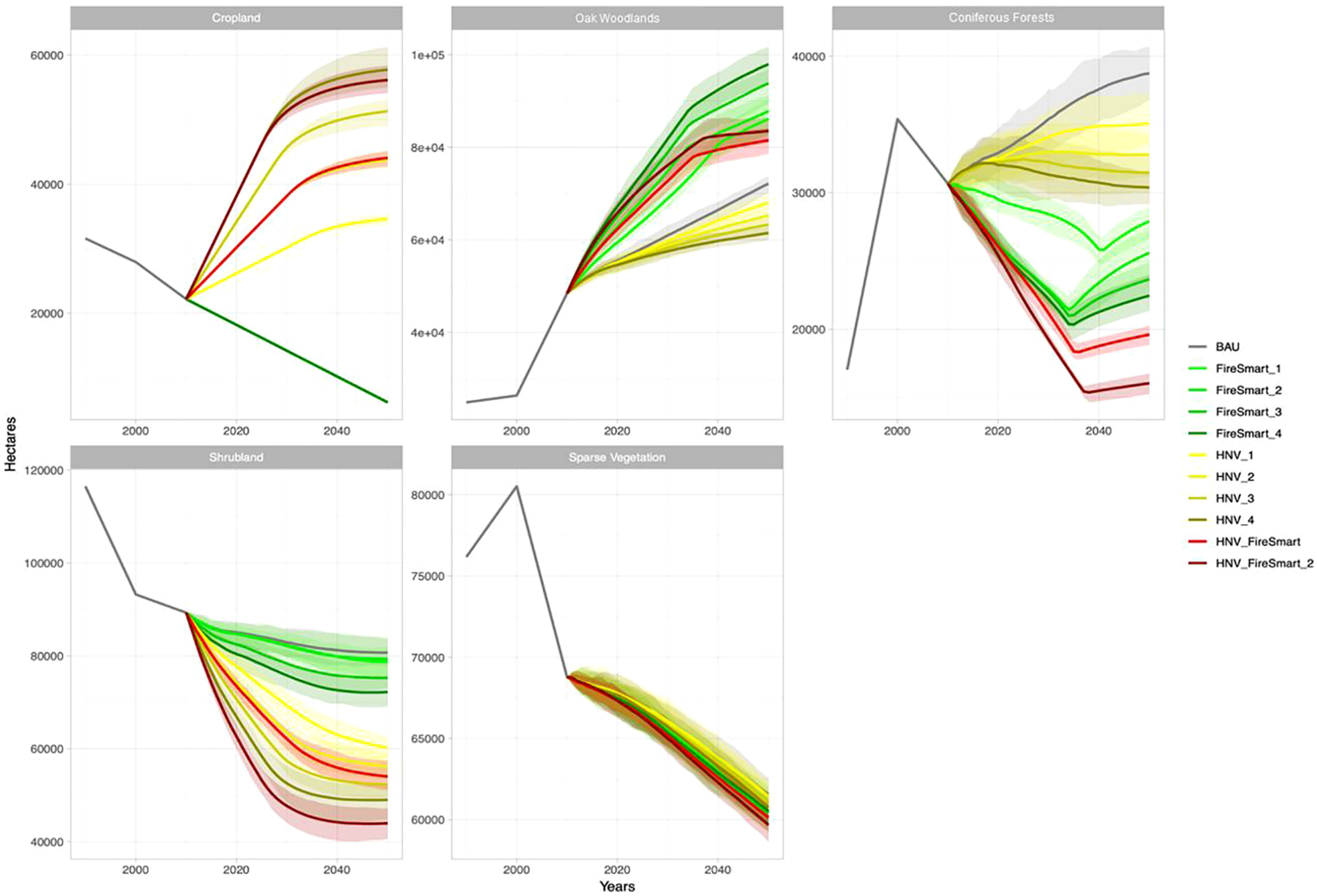

Fig. B4. Amount of each LC class (in Hectares) predicted from 1987 to 2050 under each management scenario. The intra-scenario variability is computed from the different values obtained from each of the model simulations (total of 100 replicates), and represents the uncertainty associated with fire stochasticicy.

Appendix C. . Detailed description of methods and results for carbon sequestration simulations under fire-smart management scenarios

The climate regulation ecosystem service (CRES)

Terrestrial ecosystems play an important role on global climate by controlling the concentrations of gases in the atmosphere, such as carbon dioxide, due to their ability to remove it from the atmosphere through plant photosynthesis process (i.e. carbon sequestration) and store it as carbon into plants biomass, litter and soil, to then release it back into the atmosphere through auto- and heterotrophic respiration or due to disturbances processes, such as fire or land cover change (Ciais et al., 2013). Carbon sequestration by terrestrial ecosystems (between 1960 and 2017) represents approximately 30% of the total carbon dioxide uptake in the earth system (Le Quéré et al., 2018). This capability of terrestrial ecosystems is an important regulating function (Petorelli et al., 2017) that enables the provision of the climate regulation ecosystem service (CRES) (Haines-Young and Potschin, 2018), which can benefit human well-being at global scale by avoiding potential damage costs related to carbon emissions and climate change impacts (Tol, 2018).

Methods

CRES assessment framework

We conducted a biophysical assessment of the climate regulation ecosystem service (CRES) by applying the InVEST (Integrated Valuation of Ecosystem Services and Tradeoffs) model (Sharp et al., 2018) to the Transboundary Biosphere Reserve Gerês-Xurés (TBR-GX) in order to evaluate the impact of land cover (LC) changes under scenarios of fire and land management on this ecosystem service over a period of 63 years (1987 – 2050). The total carbon sequestered (Tg C), i.e. the carbon sequestered by all carbon pools accumulated over time, and the total carbon sequestration rate (Tg yr^{-1}), i.e. the carbon sequestered per year by all carbon pools, were assumed as proxies of the CRES. Therefore, we compared the total carbon sequestration and sequestration rate among pathways of landscape evolution given by the simulated management scenarios, in order to assess how land cover change may affect the potential supply of the CRES in the study area between 1987 and 2050. Uncertainty in future carbon sequestration estimates was addressed by running the InVEST carbon module for 30 replicates per four decades in each of the four future LC scenarios. Therefore, a total of $n = 482$ simulations were run (1 simulation $\times$ 2 past dates $+$ 30 replicates $\times$ 4 decades $\times$ 4 future scenarios).

Model description, data collection and carbon modelling

The carbon sequestration and storage module of the InVEST model (Sharp et al., 2018) was used to perform the simulations. The carbon module

Table C1
Total carbon stored (Mg C) and average carbon density (Mg C/ha) per LULC date in the Biosphere Reserve Gerês-Xurés.

LULC date	Total carbon stored (Mg C)	Average carbon density (Mg C/ha)
1987	25.960.068,95	94,02
2000	26.637.466,45	96,48
2010	28.831.918,75	104,43

Table C2
Total carbon sequestered (Mg C) and carbon sequestration rate (Mg C/ha/yr) per LULC period in the Biosphere Reserve Gerês-Xurés.

LULC period	Total carbon sequestered (Mg C)	Sequestration rate (Mg C/ha/yr)
1987–2000	679.494,09	0,19
2000–2010	2.194.451,66	0,79

links the carbon stocks in four carbon pools above- and belowground biomass (AGB and BGB, respectively), litter (DOM) and soil organic carbon (SOC) to each land cover class type available in the study area, returning the carbon stored in the landscape, and compares levels of carbon over time based on LC spatial data to estimate the carbon sequestered. LC spatial databases available of the TBR-GX for the years 1987, 2000 and 2010 (30 m spatial resolution, Table C1) and the simulated landscape scenarios (2020 – 2050) classified in five major LC classes (i.e., crops, shrubs, pines, oaks, and rocky) were used to feed spatial requirements of the carbon storage and sequestration module of the InVEST model. Carbon data on AGB and BGB, DOM and SOC for each of the major LC classes was collected from data available in published scientific literature at local or regional scale, and in the official statistics from the Portuguese and Spanish national forestry inventories (Table C2) and used to estimate the carbon stocks in each of these pools per LC class required by InVEST carbon module (Table C3).

The carbon stocks in AGB and BGB in forest cover classes were computed based on the application of biomass allometric equations (Montero et al., 2005) to estimate the biomass available for the species occurring within the area, and then converted into carbon through applying a carbon content factor (Montero et al., 2005) as shown in Table S2. In addition, data on carbon in AGB available from the fifth Portuguese national forest inventory (IFN5) was directly used after applying a conversion factor (from CO_2 equivalent to C: 12 kg C/44 kg CO_2 = 0.2727). Carbon stocks in each carbon pool for all the LC classes were maintained constant over time (assuming that that carbon pools are in a steady state), which means that the carbon sequestration or emission only occurs when a pixel of a given land cover class type change between dates, while if the land cover class type is kept unchanged between dates, the carbon sequestration/emission rate will be zero for that time period.

Table C3
Sources and procedures for the estimation of carbon content in each carbon pool (AGB, BGB, DOM, and SOC) per LULC class.

LULC class	Stand data	Above-ground biomass (AGB)	Below-ground biomass (BGB)	Estimation method % of carbon in biomass	Dead organic matter (DOM)	Soil
Cropland	-	Estimates from Silva et al. (2006)	Estimates from Silva et al. (2006)	50% (Penman et al. 2003)	-	Estimates from Madeira et al. (2004)
Shrubland	-	Estimates from Viana et al. (2013)	Estimates from Viana et al. (2013)	50% (Penman et al. 2003)	Estimates from Silva et al. (2006)	Estimates from Madeira et al. (2004)
Coniferous Forests	Proença (2009) Portuguese National Forest Inventory (IFN4 - 1995 and IFN5 - 2005) Spanish National Forest Inventory (IFN2 - 1997 and IFN3 - 2005	Tree stand above-ground biomass was estimated based on the biomass allometric equations of Montero et al. (2005). Understory vegetation biomass was based on estimates from Silva et al. (2006).	Tree stand below-ground biomass was estimated based on the biomass allometric equations of Montero et al. (2005)	51.1% for P. pinaster (Montero et al. 2005)	Estimates from Silva et al. (2006)	Estimates from Madeira et al. (2004)
Oak Woodlands	Proença (2009) Portuguese National Forest Inventory (IFN4 - 1995 and IFN5 - 2005) Spanish National Forest Inventory (IFN2 - 1997 and IFN3 - 2005	Tree stand above-ground biomass was estimated based on the biomass allometric equations of Montero et al. (2005). Understory vegetation biomass was based on estimates from Silva et al. (2006).	Tree stand below-ground biomass was estimated based on the biomass allometric equations of Montero et al. (2005)	47.5% for Q. pyrenaica and 48.4% for Q. robur (Montero et al. 2005)	Estimates from Diaz-Maroto& Villa-Lameiro (2006)	Estimates from Madeira et al. (2004)
Sparsely vegetated areas	-	Assuming 30% of the estimates from Viana et al. (2013)	Assuming 30% of the estimates from Viana et al. (2013)	50% (Penman et al. 2003)	Assuming 30% of the estimates from Silva et al. (2006)	Assuming 30% of the estimates from Madeira et al. (2004)

Results

The land cover changes (observed and simulated) occurred in the Transboundary Biosphere Reserve Gerês-Xurés (TBR-GX) influenced the supply of the climate regulation ecosystem service (CRES) over time (Fig. C.1). Between 1987 and 2010 was sequestered a total of 2.87 Tg C, at an average rate of 0.12 Tg C yr^{-1} (Table S2.1 and S2.2). Regarding the future scenarios, the results indicate that the *BAU* and the *Firesmart_4* scenarios present the highest estimates for the total carbon sequestered (3.63 ± 0.27 and 4.79 ± 0.23 Tg C, respectively) and for the carbon sequestration rate (0.36 ± 0.03 and 0.48 ± 0.02 Tg C yr^{-1}, respectively), while the *HNV_FireSmart_2* and the *HNV_4* scenarios present the lowest total carbon sequestered (1.23 ± 0.17 and 0.27 ± 0.13 Tg C, respectively) and the carbon sequestration rate (0.12 ± 0.02 and 0.03 ± 0.01 Tg C yr^{-1}, respectively).Fig. C2

Considering the full period of analysis (1987 – 2050), the results indicate that the GX-BR landscape will keep supplying the climate regulation ecosystem service (CRES) in the future. However, there were differences among the scenarios (Fig. 4), which may indicate potential trade-offs between the fire and land management scenarios and the supply of the ACCRES in the area. Therefore, a higher supply of the CRES is expected if in the future the TBR-GX landscape follows the *Firesmart_4* scenario (7.66 ± 0.23 Tg C and 0.12 ± 0.004 Tg C yr^{-1}) and the *BAU* scenario (6.50 ± 0.27 Tg C and 0.10 ± 0.004 Tg C yr^{-1}), while the HNVf_FireSmart_2 (4.10 ± 0.17 Tg C and 0.065 ± 0.003 Tg C yr^{-1}) and the HNVf_4 are predicted as the less suitable scenarios for supplying the ACCRES (see Fig. 4 on the main body of the manuscript).

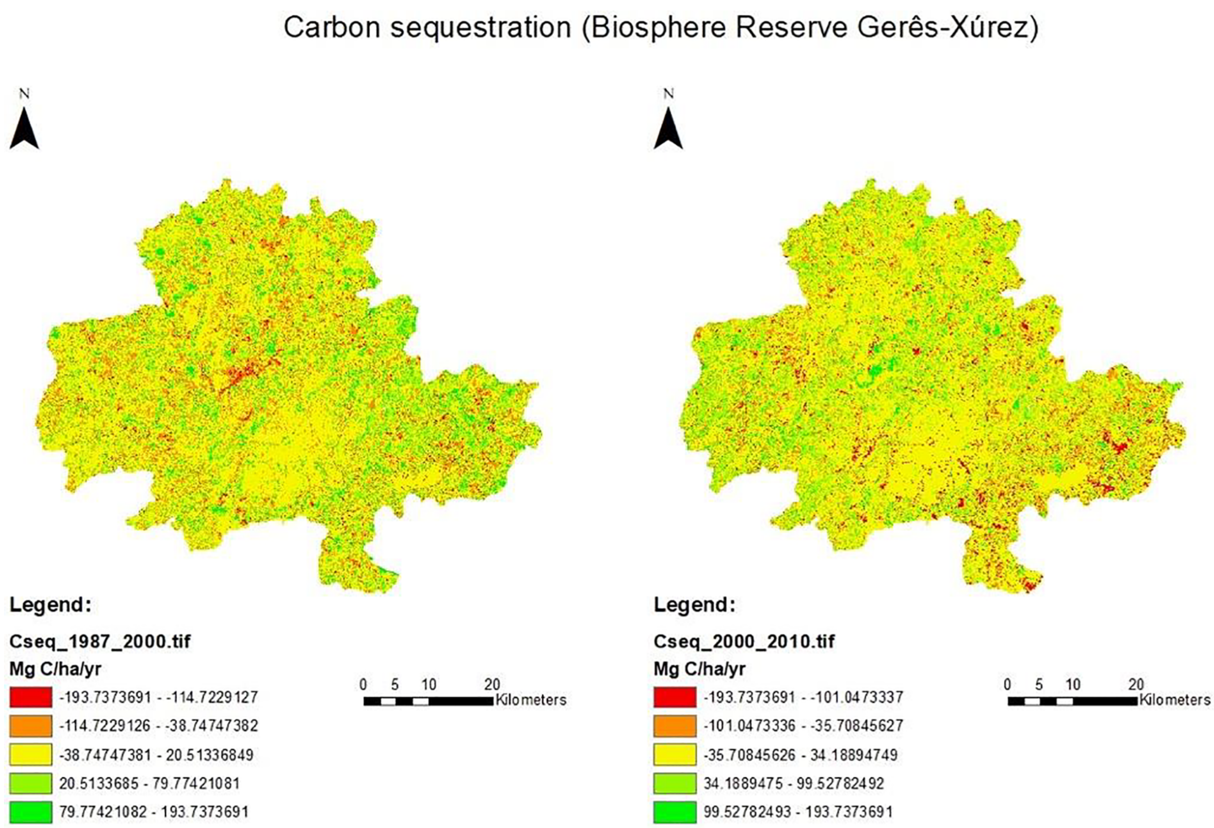

Fig. C1. Spatial representation of total carbon sequestered (Mg C) and carbon sequestration rate (Mg C/ha/yr) per LULC period in the Biosphere Reserve Gerês-Xurés.

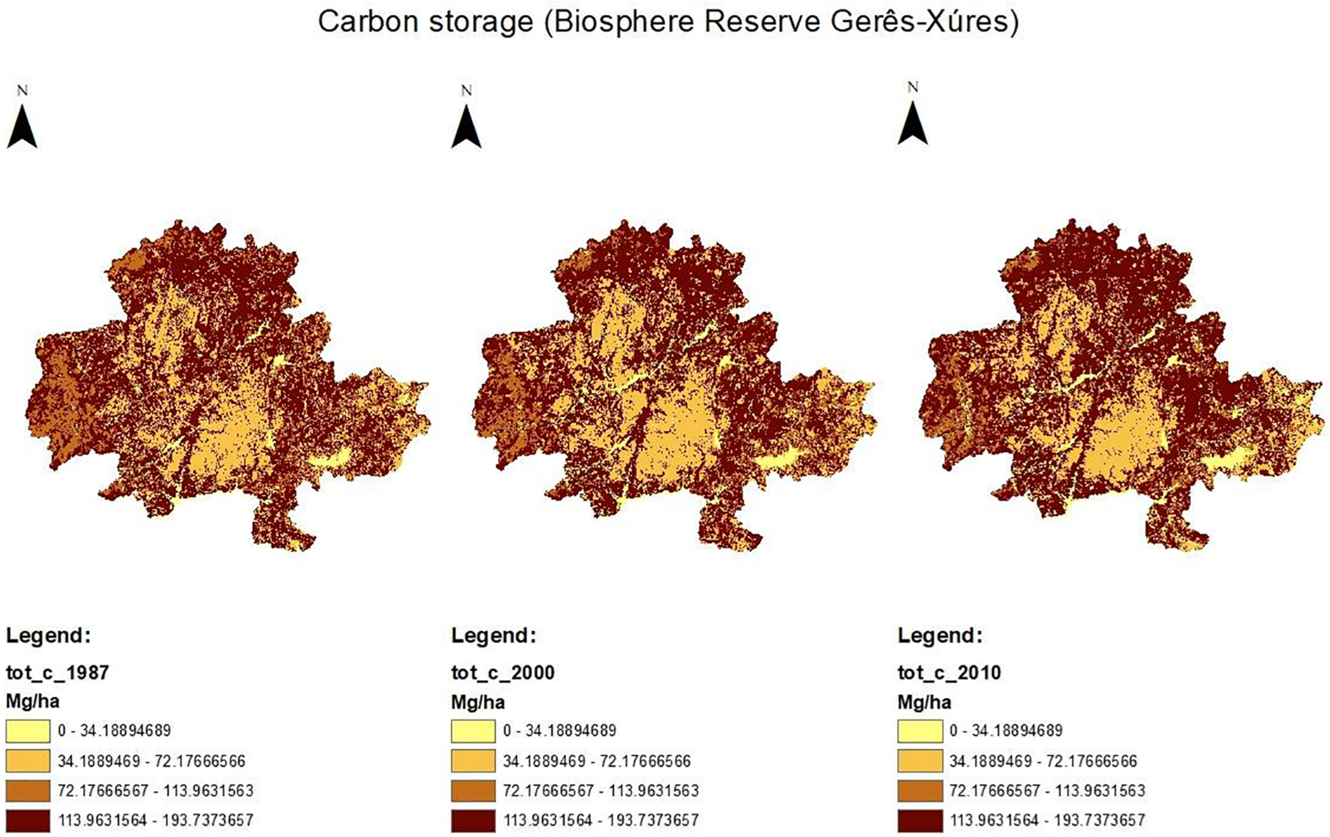

Fig. C2. Spatial representation of total carbon stored (Mg C) and average carbon density (Mg C/ha) per LULC date in the Biosphere Reserve Gerês-Xurés.

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Appendix D:. Species information and modelling criteria and evaluation metrics.

Data summary

Of 116 modeled species, 21 are under protection of the European Birds and Habitats Directives (13 bird species under the Birds Directive and four species of both amphibians and reptiles under the Habitats Directive), and 19 are considered as threatened (two amphibians, two reptile and nine bird species with Vulnerable status; one reptile and four bird species with Endangered status; and one Critically Endangered bird species) according to regional IUCN conservation criteria for Portugal and Spain. Five modeled species are endemics from Iberian Peninsula (two amphibians and three reptile species), representing unique regional biodiversity (Table D.1).

Table D1
Species information (taxonomic group, scientific name and acronyms, conservation status), modelling criteria and evaluation metrics. Model replicates refers to the number of simulations carried out for each modelling algorithm. Quality threshold indicates the value of the area under the curve (AUC) of a receiver operating characteristics (ROC) used to select models to be included on the ensemble model. AUC and TSS show the accuracy of final ensemble model.

Species information								Models			
Taxonomic group	Acronym	Scientific name	N	Birds/Habitats Directives	Regional IUCN status	Iberian endemic	Habitat	Model replicates	Quality threshold (AUC)	AUC	TSS
Amphibia	AOB	*Alytes obstetricans*	66	Yes	NT	No	Wetlands	20	0.7	0.968	0.869
Amphibia	BSP	*Bufo spinosus*	190	No	LC	No	Water/generalist	20	0.65	0.93	0.736
Amphibia	ECA	*Epidalea calamita*	78	Yes	LC	No	Wetlands	20	0.7	0.943	0.751
Amphibia	LBO	*Lissotriton boscai*	79	No	LC	Yes	Wetlands	20	0.65	0.884	0.609
Amphibia	PPE	*Pelophylax perezi*	107	No	LC	No	Water/generalist	20	0.7	0.977	0.859
Amphibia	RIB	*Rana iberica*	171	Yes	VU	Yes	Wetlands	20	0.7	0.949	0.803
Amphibia	SSA	*Salamandra salamandra*	155	No	VU	No	Wetlands/forest	20	0.7	0.906	0.66
Amphibia	TMA	*Triturus marmoratus*	64	Yes	LC	No	Wetlands	20	0.7	0.75	0.394
Aves	AAPU	*Apus apus*	175	No	LC	No	Anthropogenic	20	0,7	0.922	0.708
Aves	AARV	*Alauda arvensis*	369	No	LC	No	Scrubland	10	0,8	0.913	0.656
Aves	ACAM	*Anthus campestris*	236	Yes	LC	No	Scrubland	20	0,7	0.893	0.609
Aves	ACAU	*Aegithalos caudatus*	241	No	LC	No	Forest	20	0,7	0.890	0.638
Aves	ACHR	*Aquila chrysaetos*	45	Yes	EN	No	Mountain	10	0,8	0.987	0.917
Aves	ANOC	*Athene noctua*	22	No	LC	No	Scrubland	10	0,8	0.998	0.983
Aves	ARUF	*Alectoris rufa*	210	No	DD	No	Agricultural	20	0,7	0.876	0.589
Aves	ASPI	*Anthus spinoletta*	17	No	LC	No	Mountain	10	0,8	0.984	0.922
Aves	ATRI	*Anthus trivialis*	109	No	NT	No	Forest	10	0,8	0.950	0.763
Aves	BBUB	*Bubo bubo*	11	Yes	NT	No	Mountain	10	0,8	0.999	0.996
Aves	BBUT	*Buteo buteo*	392	No	LC	No	Forest	20	0,7	0.955	0.795
Aves	CBRA	*Certhia brachydactyla*	295	No	LC	No	Forest	20	0,7	0.875	0.610
Aves	CCAN	*Cuculus canorus*	492	No	LC	No	Agricultural	20	0,7	0.927	0.700
Aves	CCAR	*Carduelis carduelis*	39	No	LC	No	Agricultural	10	0,8	0.984	0.961
Aves	CCHL	*Carduelis chloris*	257	No	LC	No	Forest	10	0,8	0.914	0.656
Aves	CCIN	*Cinclus cinclus*	54	No	LC	No	Forest	10	0,8	0.969	0.866
Aves	CCOR	*Corvus corone*	275	No	LC	No	Agricultural	20	0,7	0.918	0.660
Aves	CCOT	*Coturnix coturnix*	180	No	DD	No	Agricultural	10	0,8	0.940	0.697
Aves	CCYA	*Circus cyaneus*	93	Yes	CR	No	Mountain	20	0,7	0.940	0.738
Aves	COEN	*Columba oenas*	15	No	DD	No	Forest	20	0,7	1	1
Aves	CORAX	*Corvus corax*	68	No	LC	No	Mountain	20	0,7	0.963	0.799
Aves	CPAL	*Columba palumbus*	329	No	LC	No	Forest	20	0,7	0.880	0.591

(*continued on next page*)

Table D1 (*continued*)

Species information								Models			
Taxonomic group	Acronym	Scientific name	N	Birds/Habitats Directives	Regional IUCN status	Iberian endemic	Habitat	Model replicates	Quality threshold (AUC)	AUC	TSS
Aves	CPYG	*Circus pygargus*	208	Yes	EN	No	Scrubland	20	0,7	0.804	0.455
Aves	DMAJ	*Dendrocopos major*	201	No	LC	No	Forest	20	0,7	0.896	0.623
Aves	DURB	*Delichon urbicum*	70	No	LC	No	Anthropogenic	10	0,8	0.967	0.890
Aves	ECIR	*Emberiza cirlus*	72	No	LC	No	Agricultural	10	0,8	0.981	0.872
Aves	ECIT	*Emberiza citrinella*	59	No	VU	No	Scrubland	10	0,8	0.981	0.862
Aves	EHOR	*Emberiza hortulana*	25	Yes	DD	No	Mountain	10	0,8	0.995	0.959
Aves	ERUB	*Erithacus rubecula*	559	No	LC	No	Agricultural	20	0,7	0.881	0.591
Aves	FCOE	*Fringilla coelebs*	543	No	LC	No	Forest	20	0,7	0.840	0.518
Aves	FPER	*Falco peregrinus*	45	Yes	VU	No	Mountain	10	0,8	0.969	0.878
Aves	FSUB	*Falco subbuteo*	39	No	VU	No	Forest	20	0,7	0.989	0.924
Aves	FTIN	*Falco tinnunculus*	138	No	LC	No	Scrubland	20	0,7	0.819	0.496
Aves	GGLA	*Garrulus glandarius*	430	No	LC	No	Forest	20	0,7	0.895	0.617
Aves	HDAU	*Cecropis daurica*	22	No	LC	No	Anthropogenic	10	0,8	0.995	0.958
Aves	HPOL	*Hippolais polyglotta*	123	No	LC	No	Scrubland	20	0,7	0.929	0.702
Aves	HRUS	*Hirundo rustica*	188	No	LC	No	Anthropogenic	10	0,8	0.917	0.675
Aves	JTOR	*Jynx torquilla*	13	No	DD	No	Forest	10	0,8	0.998	0.996
Aves	LARB	*Lullula arborea*	301	Yes	LC	No	Scrubland	20	0,7	0.880	0.606
Aves	LCAN	*Linaria cannabina*	498	No	LC	No	Agricultural	20	0,7	0.893	0.632
Aves	LCOL	*Lanius collurio*	223	Yes	NT	No	Scrubland	10	0,8	0.927	0.697
Aves	LCRI	*Lophophanes cristatus*	443	No	LC	No	Forest	20	0,7	0.942	0.751
Aves	LCUR	*Loxia curvirostra*	18	No	VU	No	Forest	10	0,8	0.982	0.845
Aves	LEXC	*Lanius excubitor*	32	No	LC	No	Scrubland	20	0,7	0.991	0.927
Aves	LMEG	*Luscinia megarhynchos*	71	No	LC	No	Scrubland	10	0,8	0.978	0.884
Aves	MALB	*Motacilla alba*	176	No	LC	No	Agricultural	10	0,8	0.948	0.761
Aves	MCAL	*Emberiza calandra*	45	No	LC	No	Agricultural	10	0,8	0.984	0.920
Aves	MCIN	*Motacilla cinerea*	115	No	LC	No	Forest	10	0,8	0.943	0.726
Aves	MFLA	*Motacilla flava*	23	No	LC	No	Mountain	10	0,8	0.992	0.951
Aves	MMIG	*Milvus migrans*	19	Yes	NT	No	Forest	10	0,8	0.998	0.991
Aves	MSAX	*Monticola saxatilis*	65	No	EN	No	Mountain	10	0,8	0.947	0.754
Aves	MSOL	*Monticola solitarius*	39	No	LC	No	Mountain	10	0,8	0.992	0.940
Aves	OHIS	*Oenanthe hispanica*	31	No	VU	No	Mountain	10	0,8	0.950	0.790
Aves	OOEN	*Oenanthe oenanthe*	74	No	LC	No	Mountain	10	0,8	0.957	0.784
Aves	OORI	*Oriolus oriolus*	170	No	LC	No	Forest	10	0,8	0.930	0.725
Aves	OSCO	*Otus scops*	55	No	DD	No	Scrubland	20	0,7	0.970	0.867
Aves	PAPI	*Pernis apivorus*	42	Yes	VU	No	Forest	10	0,8	0.803	0.490
Aves	PATE	*Periparus ater*	505	No	LC	No	Forest	20	0,7	0.880	0.592
Aves	PBON	*Phylloscopus bonelli*	159	No	LC	No	Forest	10	0,8	0.904	0.658
Aves	PCAE	*Cyanistes caeruleus*	346	No	LC	No	Forest	10	0,8	0.959	0.775
Aves	PCOL	*Phylloscopus collybita*	237	No	LC	No	Forest	20	0,7	0.912	0.663
Aves	PDOM	*Passer domesticus*	207	No	LC	No	Anthropogenic	10	0,8	0.958	0.792
Aves	PIBE	*Phylloscopus ibericus*	136	No	LC	No	Forest	20	0,7	0.956	0.773
Aves	PMAJ	*Parus major*	373	No	LC	No	Forest	10	0,8	0.880	0.604
Aves	PMOD	*Prunella modularis*	590	No	LC	No	Scrubland	20	0,7	0.881	0.630
Aves	PMON	*Passer montanus*	39	No	LC	No	Agricultural	10	0,8	0.986	0.934
Aves	POCH	*Phoenicurus ochruros*	300	No	LC	No	Anthropogenic	10	0,8	0.887	0.570
Aves	PPIC	*Pica pica*	20	No	LC	No	Agricultural	10	0,8	0.998	0.979
Aves	PPYR	*Pyrrhula pyrrhula*	232	No	LC	No	Forest	10	0,8	0.895	0.637
Aves	PRUP	*Ptyonoprogne rupestris*	45	No	LC	No	Mountain	10	0,8	0.984	0.908
Aves	PVIR	*Picus viridis*	317	No	LC	No	Forest	10	0,8	0.893	0.617
Aves	PYRR	*Pyrrhocorax pyrrhocorax*	90	Yes	EN	No	Mountain	10	0,8	0.950	0.788
Aves	RIGN	*Regulus ignicapilla*	387	No	LC	No	Forest	20	0,7	0.889	0.605
Aves	RRIP	*Riparia riparia*	30	No	LC	No	Scrubland	10	0,8	0.992	0.972
Aves	SALU	*Strix aluco*	92	No	LC	No	Forest	20	0,7	0.953	0.778
Aves	SATR	*Sylvia atricapilla*	566	No	LC	No	Scrubland	20	0,7	0.888	0.602
Aves	SBOR	*Sylvia borin*	74	No	VU	No	Forest	10	0,8	0.944	0.763
Aves	SCAN	*Sylvia cantillans*	28	No	LC	No	Scrubland	10	0,8	0.98	0.93
Aves	SCOM	*Sylvia communis*	288	No	LC	No	Scrubland	10	0,8	0.928	0.705
Aves	SEUR	*Sitta europaea*	138	No	LC	No	Forest	20	0,7	0.940	0.723
Aves	SMEL	*Sylvia melanocephala*	48	No	LC	No	Scrubland	10	0,8	0.976	0.869
Aves	SRUB	*Saxicola rubetra*	17	No	VU	No	Mountain	10	0,8	0.998	0.984
Aves	SSER	*Serinus serinus*	373	No	LC	No	Agricultural	10	0,8	0.867	0.576
Aves	STOR	*Saxicola torquatus*	678	No	LC	No	Scrubland	20	0,7	0.880	0.577
Aves	STUR	*Streptopelia turtur*	245	No	VU	No	Forest	10	0,8	0.953	0.746
Aves	SUND	*Sylvia undata*	569	Yes	LC	No	Scrubland	20	0,7	0.905	0.630

(*continued on next page*)

Table D1 (*continued*)

Species information								Models			
Taxonomic group	Acronym	Scientific name	N	Birds/Habitats Directives	Regional IUCN status	Iberian endemic	Habitat	Model replicates	Quality threshold (AUC)	AUC	TSS
Aves	SUNI	*Sturnus unicolor*	97	No	LC	No	Anthropogenic	10	0,8	0.944	0.801
Aves	TALB	*Tyto alba*	11	No	LC	No	Anthropogenic	10	0,8	1	1
Aves	TMER	*Turdus merula*	687	No	LC	No	Agricultural	20	0,7	0.946	0.805
Aves	TPHI	*Turdus philomelos*	87	No	NT	No	Forest	10	0,8	0.947	0.776
Aves	TTRO	*Troglodytes troglodytes*	686	No	LC	No	Scrubland	20	0,7	0.902	0.654
Aves	TVIS	*Turdus viscivorus*	222	No	LC	No	Forest	20	0,7	0.709	0.346
Aves	UEPO	*Upupa epops*	84	No	LC	No	Agricultural	10	0,8	0.822	0.505
Reptilia	AFR	*Anguis fragilis*	110	No	LC	No	Grasslands	20	0.65	0.951	0.809
Reptilia	CAU	*Coronella austriaca*	76	Yes	VU	No	Open woodlands	20	0.7	0.886	0.614
Reptilia	CGI	*Coronella girondica*	77	No	LC	No	Generalist	20	0.7	0.932	0.733
Reptilia	CST	*Chalcides striatus*	162	No	LC	No	Grasslands	20	0.7	0.882	0.563
Reptilia	LSC	*Lacerta schreiberi*	498	Yes	NT	Yes	Shrublands	20	0.65	0.776	0.388
Reptilia	MMO	*Malpolon monspessulanus*	126	No	LC	No	Generalist	20	0.7	0.937	0.743
Reptilia	NAS	*Natrix astreptophora*	184	No	LC	No	Water/generalist	20	0.65	0.883	0.632
Reptilia	NMA	*Natrix maura*	163	No	LC	No	Water/generalist	20	0.7	0.71	0.313
Reptilia	PAL	*Psammodromus algirus*	206	No	LC	No	Generalist	10	0.8	0.933	0.725
Reptilia	PBO	*Podarcis bocagei*	418	No	LC	Yes	Rocky/generalist	20	0.7	0.969	0.826
Reptilia	PGU	*Podarcis guadarramae*	296	No	LC	Yes	Rocky/generalist	20	0.7	0.964	0.813
Reptilia	TLE	*Timon lepidus*	387	Yes	LC	No	Shrublands	20	0.7	0.918	0.679
Reptilia	VLA	*Vipera latastei*	102	No	VU	No	Rocky/shrublands	10	0.8	0.961	0.807
Reptilia	VSE	*Vipera seoanei*	64	Yes	EN	No	Rocky/shrublands	10	0.8	0.973	0.85
Reptilia	ZSC	*Zamenis scalaris*	41	No	LC	No	Open woodlands	10	0.8	0.981	0.912

Appendix E. . Simulated burned area represented by three fire-size classes under each management scenario between 2011 and 2050.

The results showed that policies promoting farmland and cropland areas (i.e. HNVf scenarios) would lead to a significant reduction of the burned area in relation to the business-as-usual scenario (see ‘BAU’ and 'HNVf_4′ in Fig. E.1). According to our simulations, the area to be burned by large fires (> 1,000 hectares) could be potentially reduced from 20,000 hectares under the ‘BAU’ scenario up to 10,000 ha under the ‘HNF_4′ scenarios (i.e. reduction of 50%; see ‘LF’ between 2031 and 2050 in Fig. E.1). However, the reduction in the area burned by small fires (< 500 hectares) would only range from approx. 150,000 to 130,000 hectares (i.e. reduction of 13%, see ‘SF’ between 2031 and 2050 in Fig. E.1).

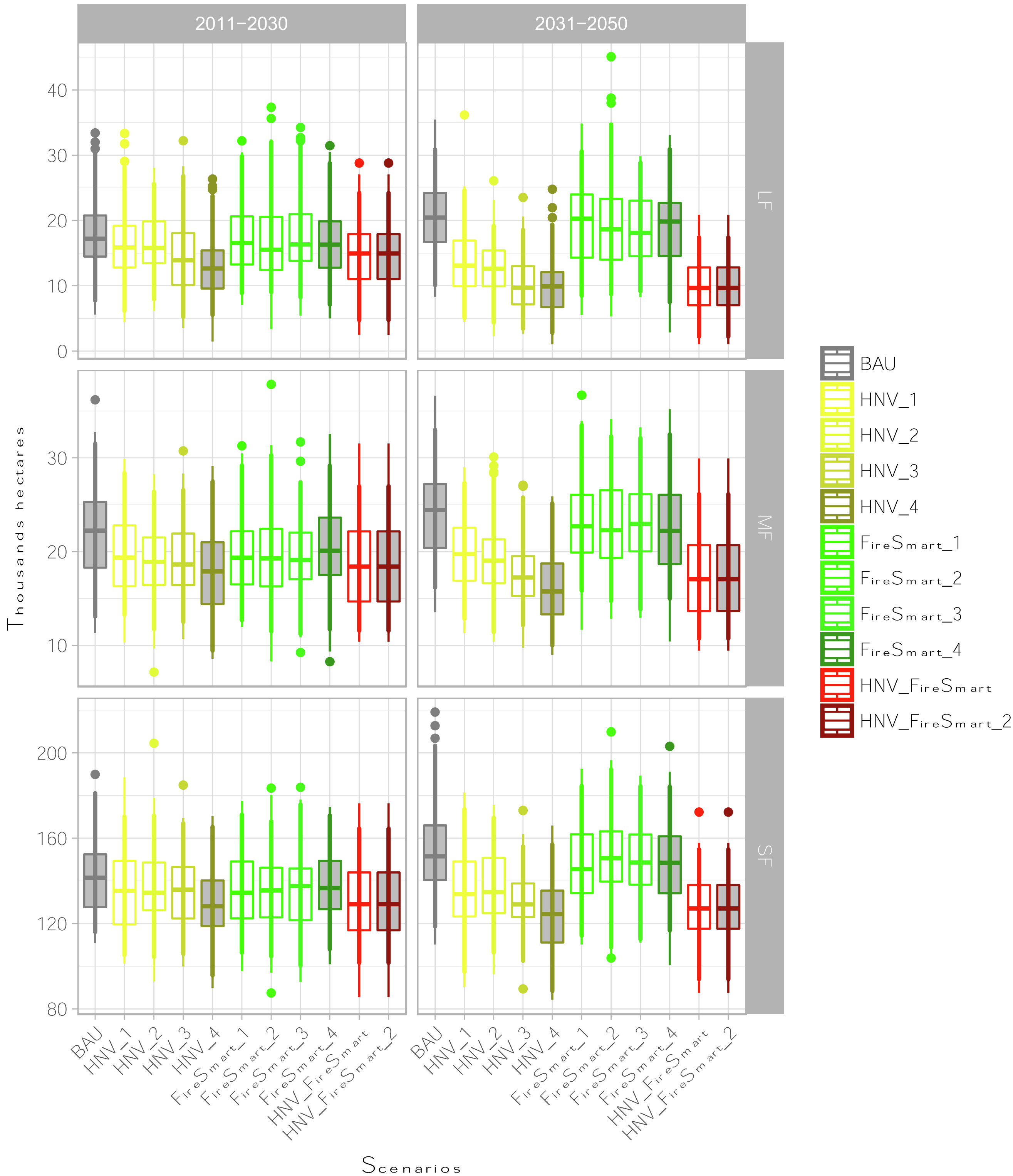

Fig. E1. Burned area (expressed in thousands of hectares) grouped by fire-size classes under each management scenario (see acronyms in Table 2) between 2011 and 2050. Results are presented for three fire-size classes: large fires (LF), medium fires (MF), and small fires (SF).

Appendix F. . Changes in habitat availability for vertebrates under different management scenarios according to the taxonomic group and conservation status.

This appendix shows the predicted changes in habitat availability for vertebrates under different management scenarios according to different conservation criteria (expressed in % of change in relation to 2010; Fig. F.1), taxonomic group (expressed in km^2; Fig. F.2) and habitat preferences for bird species (expressed in km^2; Fig. F.3). Our models predicted a wide range of contrasting species responses to management scenarios, with marked differences among taxonomic groups (Fig. F.2). According to our model projections, habitat availability for birds and reptiles would slightly

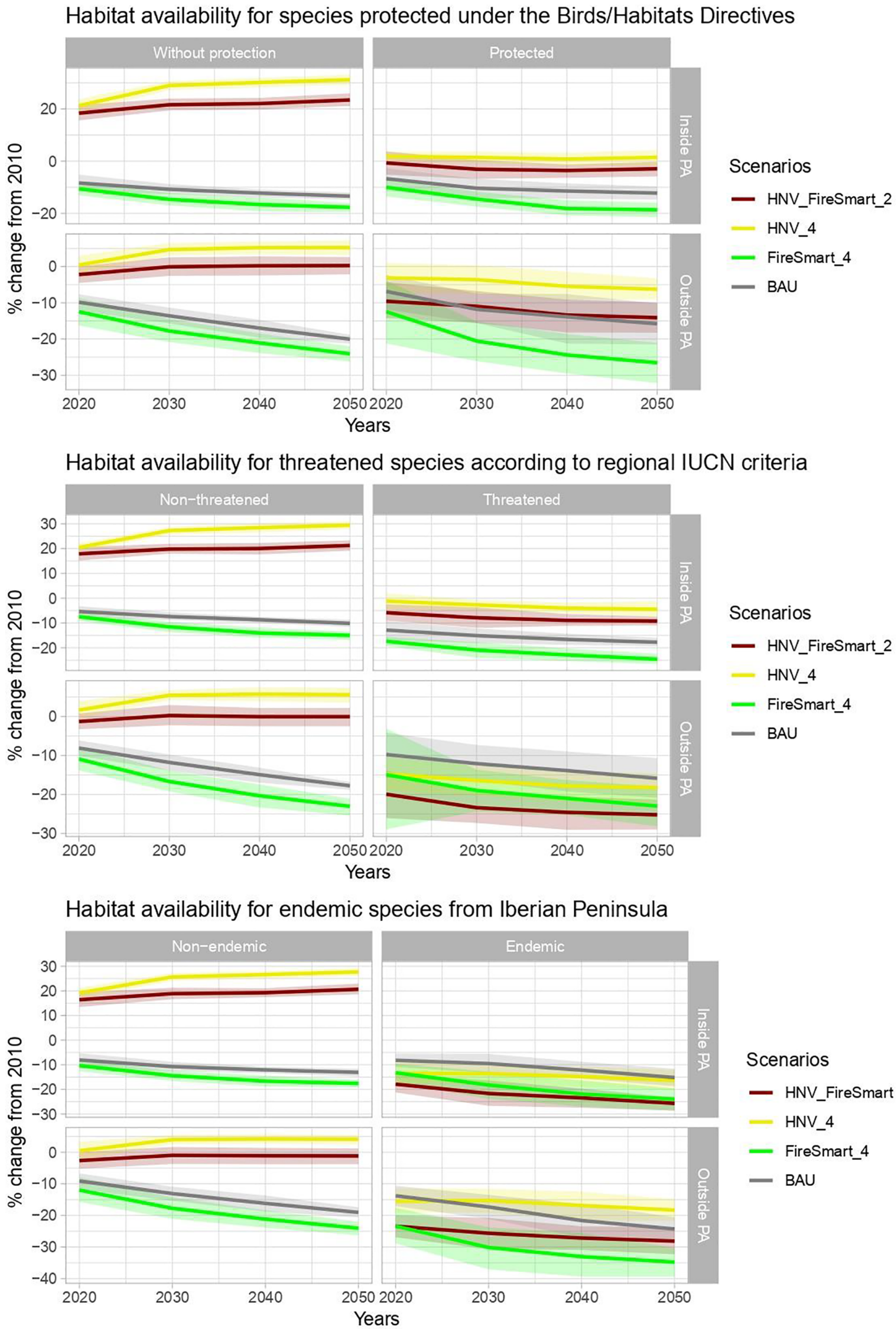

Fig. F1. Habitat availability (% of change in relation to 2010) for vertebrate species with and without protection status under different management scenarios inside and outside protected areas (see scenario acronyms in Table 2). For all plots, colored lines indicate mean values while the transparent colored areas indicate the error limits defined by the median range values. Two protection criteria are considered: the protection under the Birds and Habitats (for amphibians and reptiles) European directives (top) and the regional IUCN conservation status in Portugal and Spain (middle). For the IUCN criteria, species with status of "Least Concern" and "Near threatened" are grouped as non-threatened, while species with status of Vulnerable, Endangered and Critically Endangered are grouped as threatened. Endemic and non-endemic vertebrates from Iberian Peninsula are also represented (bottom).

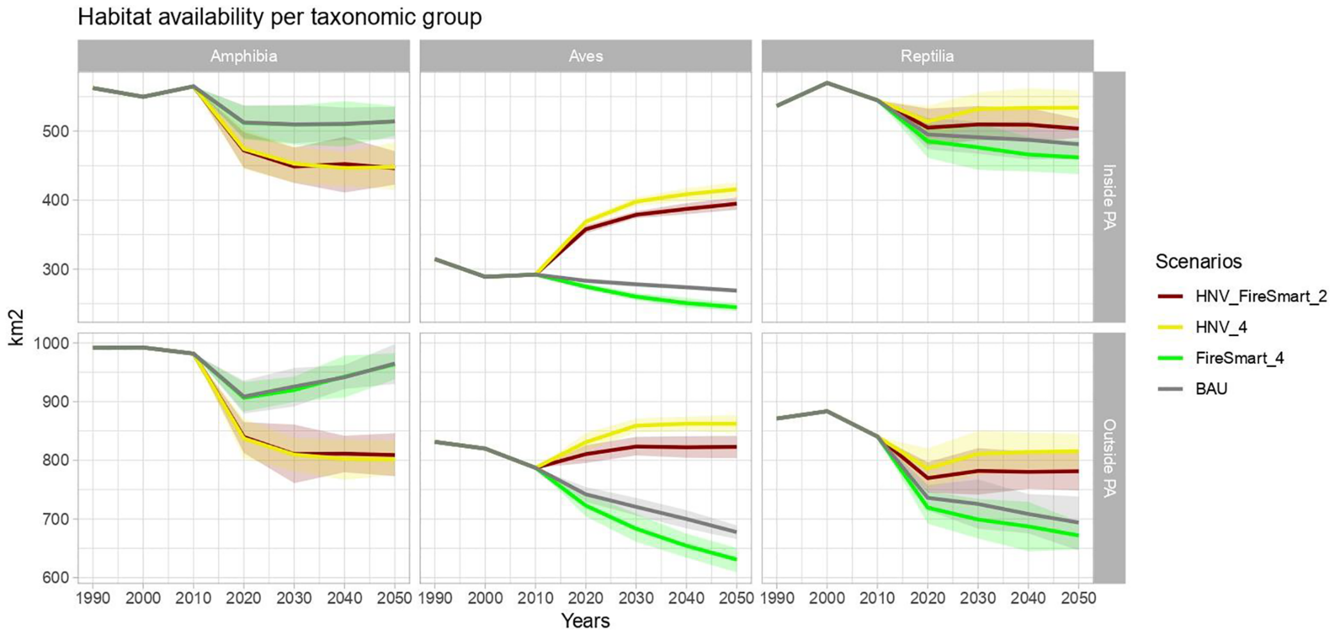

Fig. F2. Habitat availability (km^2) for amphibians, birds and reptiles inside and outside the protected areas. For all plots, colored lines indicate mean values while the transparent colored areas indicate the error limits defined by the median range values.

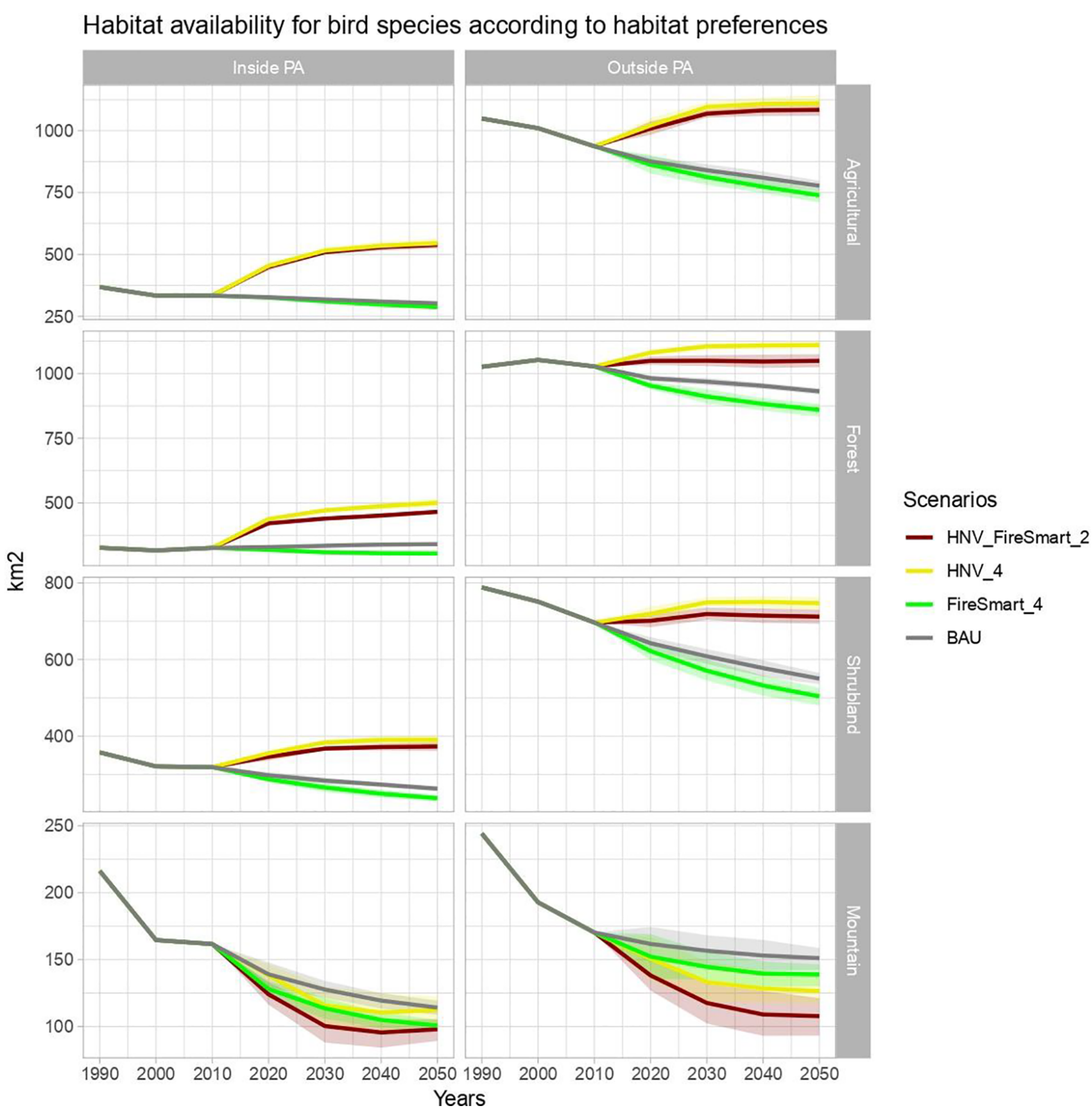

Fig. F3. Habitat availability (km2) for bird species, grouped according to respective habitat preferences, inside and outside the protected areas. For all plots, colored lines indicate mean values while the transparent colored areas indicate the error limits defined by the median range values.

decrease under the business-as-usual and fire smart scenarios (see 'BAU' and 'Firesmart' scenarios in Figs. F.2 and F.3). Policies aimed at promoting agricultural areas would favor both taxonomic groups, progressively recovering the initial values of 2010 (see 'HNVf' scenario in Figs. F.2 and F.3). However, our model projections reveal considerable losses of habitat availability for amphibian species under all management scenarios. Nonetheless, contrarily to the other taxa, the business-as-usual and fire smart scenarios could be the best option to ensure long-term habitat availability for amphibians. Outside protected areas, habitat availability is predicted to decrease considerably for all taxonomic groups.

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